Fall 2022

AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE

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Recommended Citation

Buckardt, Emma M., "AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE" (2022). *Electronic Theses & Dissertations*. 399.
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A Thesis Submitted to the Graduate School in Partial Fulfillment of the Requirements for the Degree of Master of Science

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December 2022

AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE

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ACKNOWLEDGEMENTS

I want to thank the Kansas Department of Wildlife and Parks (KDWP) State Wildlife Grants program and the United States Fish and Wildlife Service for provided the funding support for this research. I would like to thank the landowners for allowing me to survey on their properties for this project, including: KWDP, the Southeast Kansas Biological Station, Pittsburg High School (USD 250), Drs. Cindy and Steve Ford, and Andra and Brad Stefanoni. This research would not have been possible without the help of research assistants and volunteers: Austin Abrams, Ashlyn Henderson, Sara Scholes, and Gizelle Sisson.

I want to thank my family for nurturing the passion of wildlife and the outdoors at a very young age and supporting me through the research and writing of this thesis. Thank you to my sister Anna for helping me think through all parts of this project from the beginning to the end and for being there when I needed someone to know there were stampeding cows nearby. I want to thank the other graduate students and friends that I have made who built a community that supports each other in work and play, especially during the pandemic.

Thank you to my committee member Catherine Hooey and my advisors Christine Brodsky and Andrew George for their endless support and encouragement as I progressed through the project. Also thank you to Andy and Christine for helping me eat all the baked goods I made, I would not have been able to eat them all myself. This thesis is dedicated to my friends and family.

AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE

An Abstract of the Thesis by
Emma M. Buckardt

Amphibian populations are declining globally, with habitat loss and fragmentation being a leading cause for their decline. Anthropogenic changes to a landscape, such as urbanization, agriculture, and surface mining, leave few native habitats intact, which can influence amphibian populations and communities to varying degrees. Amphibians can provide insight into the health of ecosystems because they are sensitive to changes in their environment. Thus, they can be considered indicator species in anthropogenically altered wetlands. The goal of this study was to characterize amphibian communities that are using surface mined lands that have undergone vegetative succession. For Chapter I, we used call surveys to model occupancy of four anuran species, two of which are species in need of conservation (SINC; crawfish frog [*Lithobates areolatus*] and spring peeper [*Pseudacris crucifer*]). We found that anthropogenic landscape features, such as the percent of open water and cropland land cover, provided the necessary habitat to support the anuran community. In Chapter II, we evaluated the wetland characteristics that influenced the occupancy of five focal larval anuran species and the species richness and diversity of the amphibian community. We captured ten species of amphibians, including the first county record of eastern newt (*Notophthalmus viridescens*), a SINC species. Although our findings varied for each species, the change in wetland area, presence of predatory fish, water conductivity level, and percent of emergent vegetative cover explained the variation in occupancy patterns for most species and for the amphibian community within a wetland. We also found that larval amphibian communities did not differ between management or land use history of the site. Lastly in Chapter III, we assessed the efficacy of survey methodology on the capture rates of larval amphibians. We found that baiting minnow traps with green glowsticks increased capture rates overall, but this effect was species-specific and varied by the time of year. The findings from all three studies provide important insights regarding amphibian use of formerly mined landscapes. The factors that determine species occupancy and community structure are related to both landscape composition and local habitat features, regardless of land-use history. Even sites that have been heavily disturbed by surface mining can potentially provide habitat for multiple amphibian species, including at-risk species. The conservation value of these recovering wetlands warrants their management and protection.

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CHAPTER I

PATTERNS OF ANURAN OCCUPANCY ON A POST-MINED LANDSCAPE

ABSTRACT

Anuran populations are declining globally, with habitat loss and fragmentation being a leading cause for their decline. Anthropogenic changes to a landscape, such as urbanization, agriculture, and surface mining, leave few native habitats intact, which can influence anuran populations and communities to varying degrees. Our study aimed to assess the connection between anuran occupancy and anthropogenic and native habitats across a landscape that was heavily disturbed by surface mining and row crop agriculture. We conducted call surveys six times from mid-March to mid-June in 2021 and 2022 at 65 sites throughout Crawford and Cherokee cos. in southeast Kansas. We conducted single-species single-season occupancy modeling for four out of nine detected anuran species, as the other species were nearly ubiquitous on the landscape. We used land cover types to model occupancy for American bullfrog (*Lithobates catesbeianus*), crawfish frog (*Lithobates areolatus*), gray treefrog (*Hyla versicolor*), and spring peeper (*Pseudacris crucifer*). We recorded nine anuran species calling, with naïve occupancy varying from 38% to 100%. American bullfrogs were positively associated with open water and built cover, while gray treefrogs had a weak association with grasslands. Crawfish frogs were positively associated with croplands and had a slightly higher occupancy in the Spring

River sub-basin in 2022. Spring peepers' occupancy was nearly exclusively within the Spring River sub-basin, and negatively associated with cropland and urbanization. The anthropogenic landscape provided the necessary habitats to support species such as the crawfish frog and spring peeper, which are species in need of conservation, as well as more ubiquitous species like the boreal chorus frog (*Pseudacris maculata*). Management of habitats within an anthropogenic landscape can support current and future anuran communities, including imperiled species.

INTRODUCTION

Amphibian populations are declining globally, including species that are locally common. For example, approximately 33% of anurans are currently considered threatened by the International Union for Conservation of Nature and Natural Resources (Adams et al., 2013; Bishop et al., 2012; IUCN, 2022). While the severity and specific mechanisms affecting amphibian populations vary across species and regions (Campbell Grant et al., 2020; Cushman, 2006; Gallant et al., 2007), anuran populations are greatly impacted by the loss of both the aquatic and terrestrial habitats used throughout their life cycle (Knutson et al., 1999). The loss and fragmentation of wetlands across the landscape has altered species composition, especially limiting species with low dispersal capabilities (Brodman, 2008; Gibbs, 2000). Human influence on the landscape has been another important cause of amphibian declines in wetland habitats to the extent that 22% wetland-dependent amphibians in North America are considered threatened by the IUCN (Ramsar Convention on Wetlands, 2018). Herein, we focus on three anthropogenic disturbances upon landscapes facing amphibian populations: urbanization, row-crop agriculture, and surface mining.

Urbanization alters wetland and upland habitats through changes at both local and landscape scales (Johnson et al., 2013). For example, housing developments can have a prolonged, detrimental effect on amphibian populations due to increased pollutant exposure and increased habitat fragmentation (Gagné & Fahrig, 2010; Johnson et al., 2013; Pillsbury & Miller, 2008). Habitat fragmentation resulting from roads and buildings further isolates aquatic habitats and upland habitats that are needed to support anurans (Eigenbrod et al., 2008; Gibbs, 2000). The increase in urban sprawl increases the abundance and density of impervious surfaces, which not only can increase road mortality, but also alter can wetlands through stormwater and pollution run-off (Beebee, 2013; Johnson et al., 2013; Smallbone et al., 2011). These local and landscape changes are particularly important as global human populations become more concentrated and urbanized landscapes expand (Seto et al., 2012), yet amphibian populations continue to be understudied in urban ecosystems (Rega-Brodsky et al., 2022).

Agricultural practices also may negatively affect amphibian populations and communities. The degradation of native habitat, such as the removal of forest for row crops, can reduce anuran diversity and populations, especially as the increased use of agricultural pesticides can influence species survival (Cayuela et al., 2015; Smith et al., 2006). The strengths and directionality of these effects can vary by the intensity of the agricultural operation and the species studied, in some cases positively affecting anuran populations (Koumaris & Fahrig, 2016). In one study, the amount of cropland around wetlands positively influenced American toad (*Anaxyrus americanus*) and northern leopard frog (*Lithobat*es *pipens*) occupancy (Swanson et al., 2019). Other agricultural practices, like farm pond management, may provide more wetland habitat that is otherwise limited on the landscape, increasing anuran species richness and diversity (Swartz & Miller, 2021).

Current anthropogenic habitat disturbance like urbanization and agriculture greatly impacts anuran occupancy. However, the historic land use and cover of the landscape can be just as influential to current populations, creating a need to understand how past and present land cover is driving anuran occupancy (Piha et al., 2007). For example, the reclamation of past surface mining operations influences habitat quality as the reclamation process often changes the hydrology and vegetation of wetlands (Stiles et al., 2017). Through this reclamation process, additional breeding habitats may be created to support an amphibian community that is at least as diverse as natural wetlands (Fetting, 2014; Lannoo et al., 2009; Lannoo et al., 2014; Timm & Meretsky, 2004). Thus, the land use history can dictate future vegetative succession and how the anuran populations respond to the change in land cover.

Urban, agricultural, and post-mining landscapes are each impacting anuran communities through habitat loss and fragmentation. This study sought to provide a connection between anuran occupancy and landscape composition in a highly altered landscape and provide insight for anuran management. We used anuran call surveys and landscape metrics for five land cover types (i.e., water, grassland, cropland, forest, and built environment) to associate species occupancy with the landscape. We predicted that the native habitat types such as forest, water, and grasslands were the most important land cover type for anuran species in the area, as many of the species studied require these features for reproduction. In contrast. urbanization and agriculture should negatively impact anuran occupancy due to the resulting habitat changes and overall loss of wetland habitats. Understanding the association between anuran distributions and landscape composition could help inform conservation actions to support anurans in anthropogenic landscapes.

METHODS

Study Area

We sampled a study area spanning southern Crawford Co. and northern Cherokee Co. in southeast Kansas. These counties belong to the Cherokee lowland physiographic region, which is characterized by rolling plains with patches of riparian forests and revegetated former surface mining areas (Fig. 1.1; Kansas Geological Survey 1999). The eastern portion of study area is a part of the Spring River sub-basin, and the western portion is a part of the Neosho River basin (Fig. 1.1). This region was mined for coal and other metals from the 1850s to the 1980s, with most surface mining areas left unreclaimed to be naturally revegetated (Bailey & Hooey, 2017; Kansas Historical Society, 2013). The Kansas Department of Wildlife and Parks (KDWP) and the Kansas Department of Health and the Environment (KDHE) have been working to reclaim 14,500 acres of historic strip-mined areas, which are collectively known as the Mined Land Wildlife Area (MLWA; KDWP, 2018). The KDWP and KDHE have already reclaimed some of this land into grasslands and marshes to help improve habitat quality for wildlife, such as waterfowl and upland game birds. The remainder of the land cover on the MLWA is comprised of forest, shrub, and water, and is surrounded by agricultural and urban land uses.

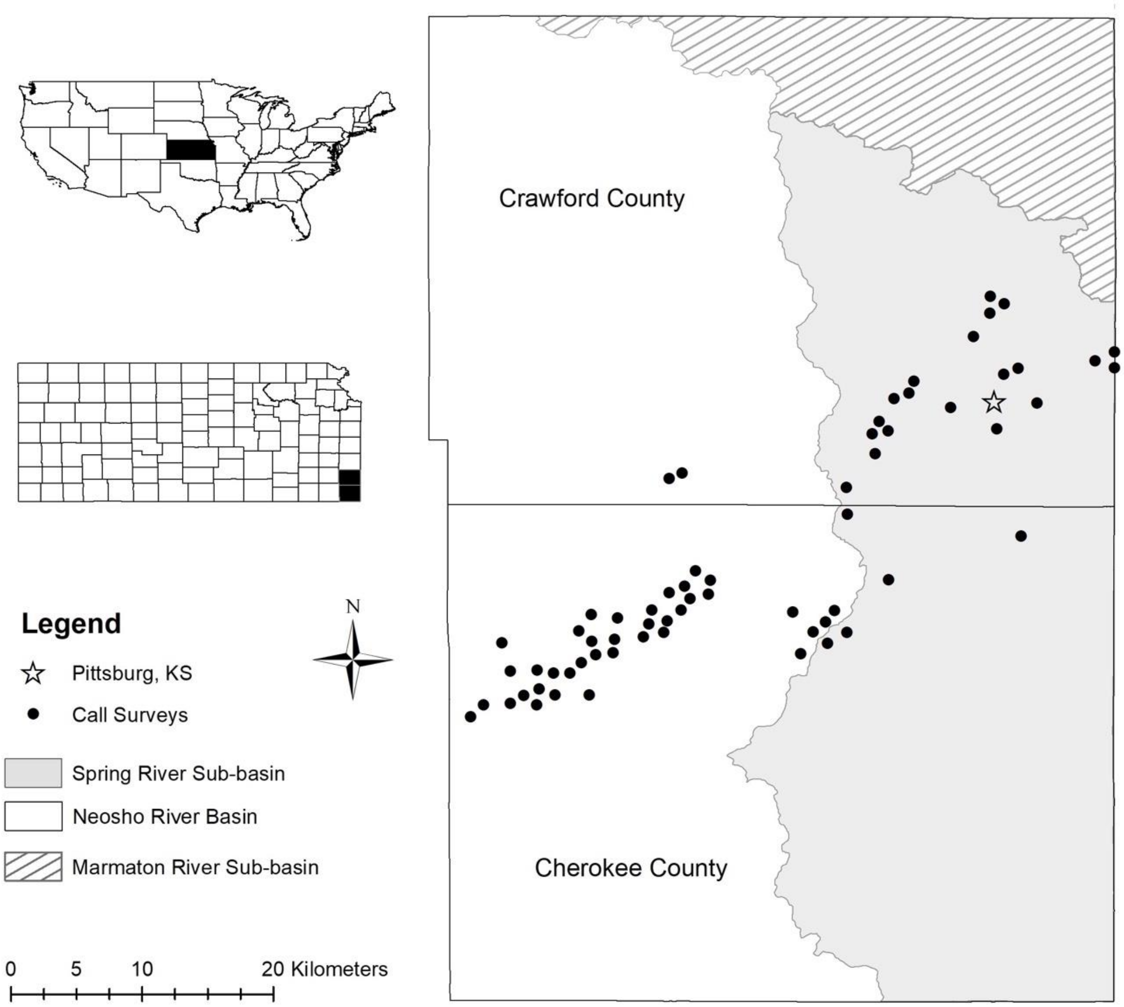

Figure 1.1. Map of the survey area with watersheds depicted. Dots represents call survey locations.

Auditory Call Surveys

We conducted anuran call surveys at 65 sites across the region on mined and non-mined lands during the 2021 and 2022 breeding seasons (Fig. 1.; Appendix I). To get the broadest coverage and ensure spatial independence of samples, we chose sites that were greater than 500 m apart and accessible from roadways or parking lots throughout the survey area, following the North American Amphibian Monitoring Program (NAAMP) protocol (Weir & Mossman, 2005). Surveys occurred twice during three different survey windows defined as mid-March to mid-April (early spring), May (spring), and June (early summer) to account for the variability of breeding times of anuran species in Kansas. Within each survey window, we sampled groups of sites in a random order and all sites were surveyed within 10 days of each other. We conducted surveys between 30 min after sunset and 0100 hrs (Weir & Mossman, 2005).

After arriving at each site, we had a 1-minute acclimation period before calls were recorded to reduce disturbance impacts on area anurans (Stevens et al., 2002). During this period, the surveyor measured detection variables including air temperature, average wind speed (Kestrel Weather Meter 2000), and percent cloud cover. Surveys were not conducted when the wind was greater than 16 kph or when there was heavy rain (Weir & Mossman, 2005). After the acclimation period, the surveyor listened for anuran calls for five minutes and recorded the strength of the chorus for every species heard (Crouch & Paton, 2002; Pierce & Gutzwiller, 2004). The surveyor determined the strength of the chorus by following index based on NAAMP: 1= individuals can be counted with a space between calls, 2= calls of individuals can be distinguished with some overlapping of calls, 3= full chorus with constant, continuous, and overlapping calls (Weir & Mossman, 2005). Surveyors also recorded the presence of other noise events (e.g. trains, passing cars, braking dogs, and people talking) that may have inhibited detection based on the NAAMP noise scale (Weir & Mossman, 2005). All surveyors were trained prior to data collection to ensure consistent and accurate aural data collection.

Land Cover Data Collection

We used the most recent National Wetland Inventory (NWI) and National Land Cover Data (NLCD) to collect proportions of land cover types within 500-m buffers around each survey location. We chose a 500-m buffer because it is considered within in the range of core habitat and average known dispersal distance of anuran species (Eigenbrod et al., 2008; Semlitsch & Bodie, 2003). To simplify land cover types, NWI was reclassified into two water body types and NLCD was reclassified into five land cover types within Program R, version 1.3.1073 (Table 1.1;R Core Team, 2020). Much of the MLWA was classified as woody wetlands, but the MLWA is primary a terrestrial habitat with distinct waterbodies, instead of trees in standing water. Therefore, we reclassified woody wetlands as forest to represent the true forest cover more accurately in the area using the NLCD data and used NWI data to assess all distinct aquatic habitats that may have been lost with reclassification. We included bare ground within urban land cover since this land cover type in this region was a result of human manipulations. Grassland land cover included pastures because they likely function as grasslands in the study area for amphibian populations. We used package “raster” to obtain the class percent from the landcover data created from the reclassified NWI and NLCD data with a 500-m circular buffer (Hijmans, 2022). These percentages were z-transformed prior to use in statistical analysis.

Table 1.1. Reclassification of the National Wetland Inventory (NWI) into two water body types and the National Landcover Database (NLCD) into five landcover types (Dewitz, 2021; U.S. Fish & Wildlife Service, 2022).

Landcover Category	Cover Types included in NWI or NLCD
NWI	
Open Water	Lake
	Freshwater Pond
	Riverine
Wetland	Freshwater Emergent Wetland
	Freshwater Forested/Shrub Wetland
NLCD	
Forest	Deciduous Forest
	Evergreen Forest
	Mixed Forest
	Shrub/Scrub
	Woody Wetlands
Water	Open water
	Emergent Herbaceous Wetlands
Built	Developed, Open Space
	Developed, Medium Intensity
	Developed, High Intensity
	Barren Land
Grass	Grassland/Herbaceous
	Paster/Hay
Crop	Cultivated Crops

Data analysis

We used single-season occupancy models to determine how landscape composition and wetland types around the survey point affected occupancy for each species using the package “unmarked” (Fiske & Chandler, 2011; Weir et al., 2005, 2014; Weir & Mossman, 2005). Species detected on >90% of sites were excluded from analyses because they lacked sufficient variability for modeling occupancy. Before fitting models, we tested covariates for multicollinearity and only included variables with $r < 0.7$

within the same models. The sampling window for each species reflected their average call phenology window in Kansas (Taggart, 2022).

We fitted models using presence-absence data in a stepwise process starting with detection probability using detection covariates. We used an additive approach to determine one or two variables that influence detection probability. We used Akaike's information criterion corrected for small sample size (AIC_c) to determine which models were supported by the data ($\Delta AIC_c<2$; Hurvich & Tsai, 1989). Models for detection that were supported were used in modeling for occupancy. We then created models that estimated the probability of occupancy using the occupancy covariates (Table 1.2). We used an additive approach to determine one to three influential variables based on AIC_c and model weight. We tested for overdispersion and examined goodness of fit to assess the overall fit of the best model (MacKenzie & Bailey, 2004). We concluded the modeling procedure for gray treefrog after the addition of one occupancy variable due to the lack of convergence.

Table 1.2. Variables used as detection and occupancy covariates in occupancy models of anuran species surveyed in southeast Kansas. All occupancy variables represent measurements from a 500-m buffer around the survey point except watershed, which was based on the specific survey point.

Model Parameter	Description
Detection Covariates	
day	The ordinal date of survey
year	Survey year: 2021, 2022
time	Minutes past sunset, calculated as the difference in sunset time and survey start time
cloud	Estimated percent of cloud cover at time and site of survey
noise	Ambient noise level based on NAAMP index
obs	Observer conducting the survey
Occupancy Covariates	
year	Survey year: 2021,2022
watershed	Watershed survey occurred in (Spring River sub-basin or Neosho River basin)
water	Proportion of open water, based on National Wetland Inventory (1985)
wetland	Proportion of wetlands, based on National Wetland Inventory (1985)
forest	Proportion of forest, based on National Land Cover Database (2019)
crop	Proportion of cropland, based on National Land Cover Database (2019)
grass	Proportion of grassland, based on National Land Cover Database (2019)
built	Proportion of built environment, based on National Land Cover Database (2019)

RESULTS

Our surveys resulted in the detection of nine anuran species: Blanchard's cricket frog (*Acris blanchardi*, detected at 100% of surveyed sites), American toad (98%), boreal chorus frog (*Pseudacris maculata*, 98%), Cope's gray treefrog (*Hyla chrysoscelis*, 98%), southern leopard frog (*Lithobates sphenocephalus*, 98%), American bullfrog (*Lithobates catesbeianus*, 89%), spring peeper (46%), crawfish frog (40%), gray treefrog (*Hyla versicolor*, 38%; Appendix II). Detection probability for all four modeled anuran species included ordinal day along with an addition variable (Table 1.3). Detection probability of American bullfrog, crawfish frog, and gray treefrog was also influenced by the observer, while detection probability of spring peepers was also explained by the ambient noise levels (Table 1.3).

The best supported model for American bullfrog occupancy included the amount of open water and built environment; open water and the built environment increased with the likelihood of occupancy (Table 1.3; Fig. 1.2; Appendix III). The best supported model for crawfish frog occupancy included the proportion of cropland, watershed, and year (Table 1.3; Appendix IV). Crawfish frogs were more likely to occupy a site with greater cropland coverage within 500 m, if the site was within the Spring River sub-basin, and in the second year (Table 1.4; Fig. 1.3). The best supported model for gray treefrog occupancy included the amount of grassland cover (Table 1.3; Appendix V). However, the estimated coefficient for grassland cover included zero, limiting the strength of the inferred relationship (Table 1.4; Fig. 1.4). The best supported model for spring peeper occupancy included watershed and the proportion of built environment and cropland (Table 1.3; Appendix VI). Spring peepers were more likely to occupy sites within the

Spring River sub-basin and with a small proportion of built and cropland cover types within 500 m (Table 1.4; Fig. 1.5).

The map derived from top model for crawfish frogs predicted occurrence throughout a large percentage of the survey area (Fig. 1.6B). The areas with limited occurrence were primarily tied to the MLWA and urban centers like Pittsburg, KS. The map derived from the top model for spring peepers predicted occurrence was more limited across the survey area (Fig. 1.6C). The areas with the highest likelihood of occurrence were primarily within historical surface mined areas through the center of the survey area.

Table 1.3. Top-ranked (ΔAIC_c <2) occupancy models estimating the probability that American bullfrogs (*Lithobates catesbeianus*), crawfish frogs (*Lithobates areolatus*), gray treefrogs (*Hyla versicolor*), and spring peepers (*Pseudacris crucifer*) occupied a call site during 2021 and 2022 in southeast Kansas. Null models have also been included, along with each models' parameters (K) and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
American bullfrog			
p(day + obs) ψ(water + built)	8	0	0.92
p(day + obs) ψ(.)	6	32.35	0
Crawfish frog			
p(day + obs) ψ(crop + watershed + year)	9	0	0.65
p(day + obs) ψ(.)	6	9.23	0.01
Gray treefrog			
p(day + obs) ψ(grass)	7	0	0.30
p(day + obs) ψ(built)	7	0.42	0.24
p(day + obs) ψ(wetland)	7	1.65	0.13
p(day + obs) ψ(.)	6	2.76	0.08
Spring peepers			
p(day + noise) ψ(watershed + built + crop)	10	0	0.83
p(day + noise) ψ(.)	7	97.95	0

Table 1.4. Estimates of each occupancy parameter with the top occupancy model for American bullfrogs (*Lithobates catesbeianus*), crawfish frogs (*Lithobates areolatus*), gray treefrogs (*Hyla versicolor*), and spring peepers (*Pseudacris crucifer*), based on call surveys conducted in 2021 and 2021 in southeast Kansas. The beta estimates, standard errors (SE), and the lower and upper 95% confident interval (CI) for each parameter were included.

Species	Parameters	Estimate	SE	Lower 95% CI	Upper 95% CI
American bullfrog	intercept	0.39	0.64	-0.67	1.45
	water	1.58	0.74	0.37	2.79
	built	-0.52	0.31	-1.02	-0.01
Crawfish frog	intercept	-3.29	3.22	-8.59	2.02
	crop	2.44	1.63	-0.24	5.11
	watershed	6.33	62.01	-95.67	108.33
	year	4.00	2.92	-0.81	8.81
Gray treefrog	intercept	12.90	15.00	-11.81	37.57
	grass	10.70	11.50	-8.18	29.64
Spring peeper	intercept	-5.37	1.58	-7.97	-2.78
	watershed	7.36	1.50	4.88	9.83
	built	-1.08	0.33	-1.62	-0.53
	crop	-3.24	1.29	-5.36	-1.13

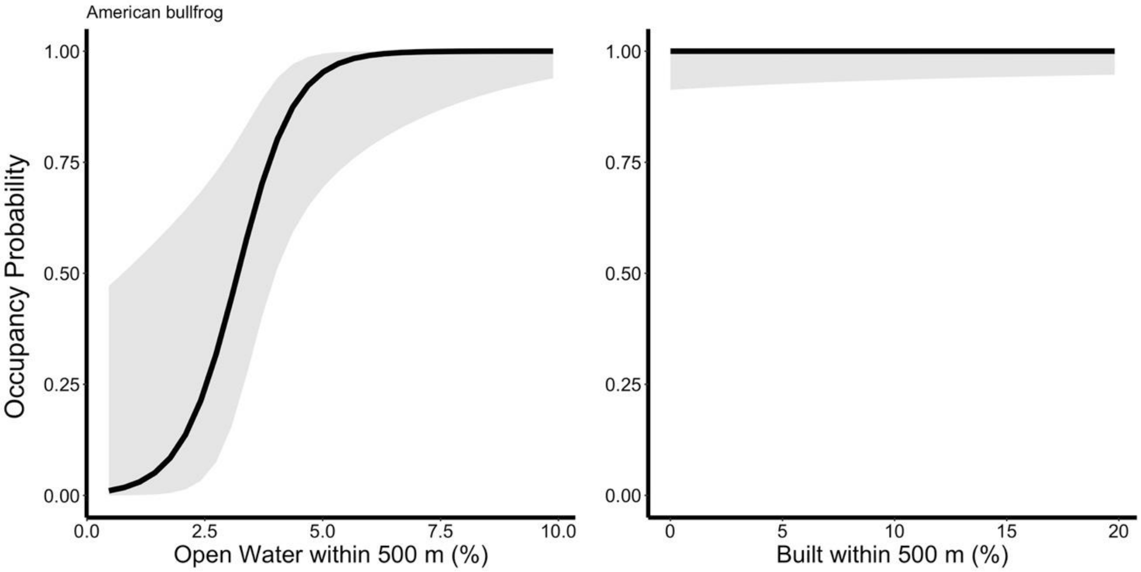

Figure 1.2. Predictive plots based on the top occupancy model for American bullfrogs (*Lithobates catesbeianus*) during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

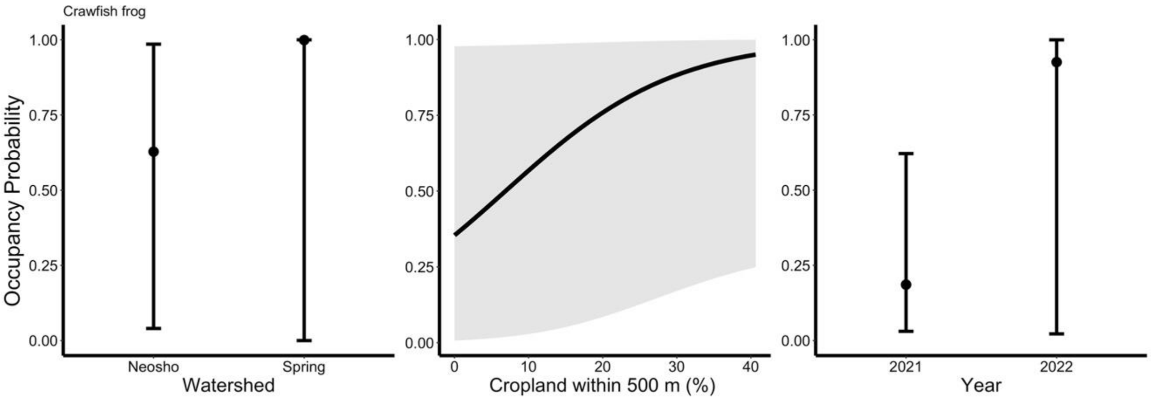

Figure 1.3. Predictive plots based on the top occupancy model for crawfish frog (*Lithobates areolatus*) during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals. Watershed was held at its intercept when making predictive plots for other variables within the top model.

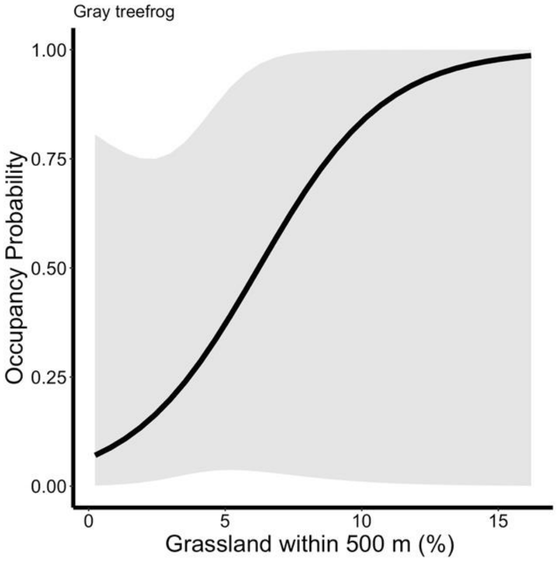

Figure 1.4. Predictive plots based on the top occupancy model for gray treefrogs (*Hyla versicolor*) during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

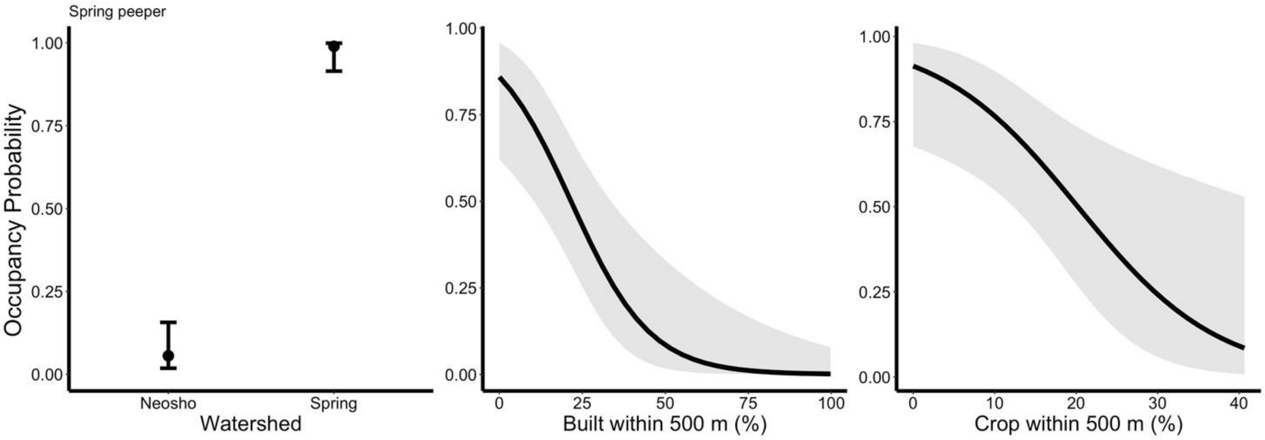

Figure 1.5. Predictive plots based on the top occupancy model for spring peeper (*Pseudacris crucifer*) during breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

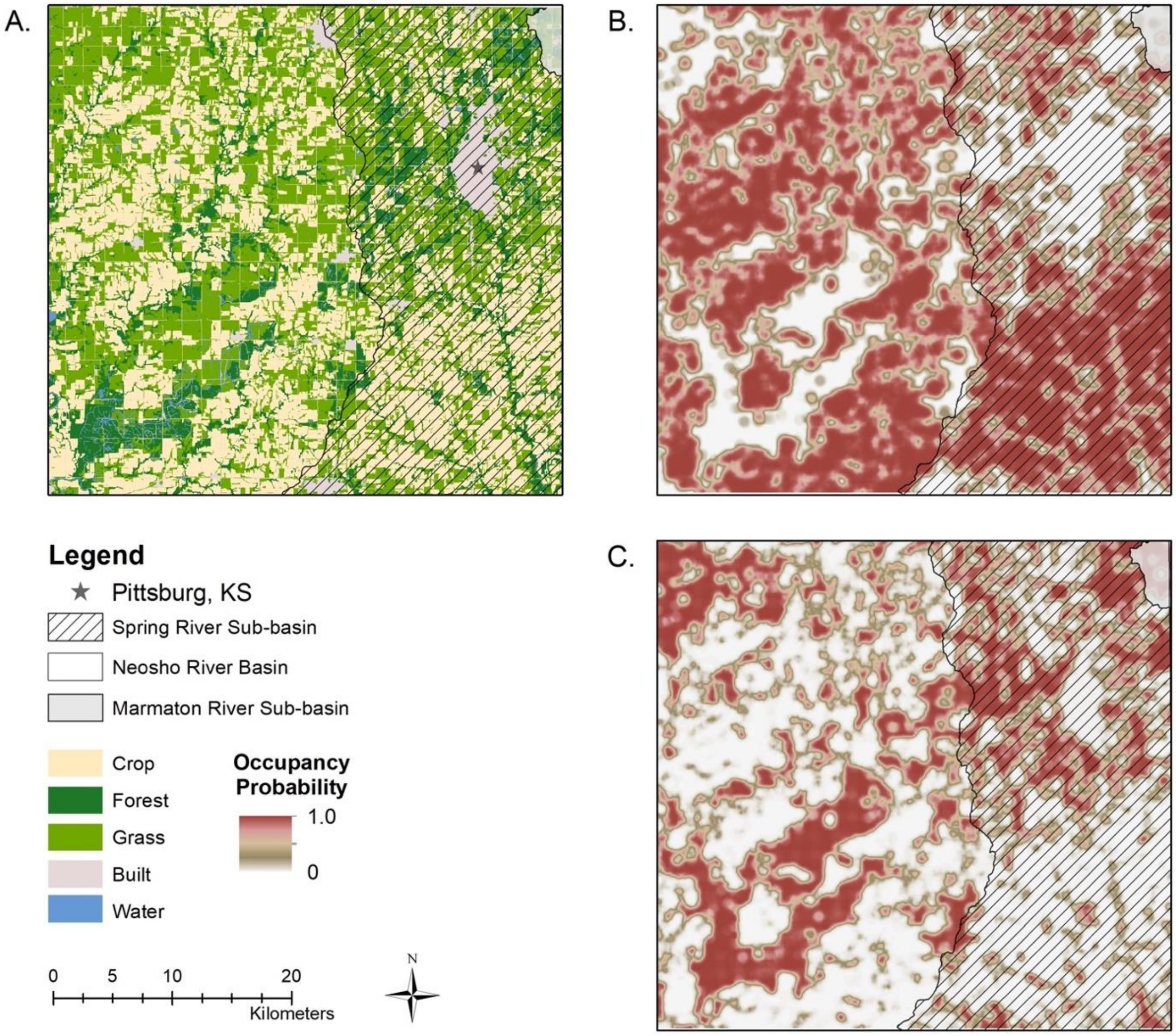

Figure 1.6. Land cover categories reclassified from the NLCD and associated watersheds in southeast Kansas A), and the resulting probability of SINC species occupancy, B) crawfish frog (*Lithobates areolatus*) and C) spring peeper (*Pseudacris crucifer*), during the breeding season in 2021 and 2022. Occupancy plots were created based on the best model for single-species occupancy models. Categorical variables were held at their respective intercepts.

DISCUSSION

In a highly fragmented and disturbed landscape, we found that occupancy of anuran species depended on the specific habitat needs of each species rather than one prevailing habitat disturbance or land cover feature. Even though the survey area has been highly impacted by anthropogenic changes like surface mining, agriculture, and urbanization, the landscape provided aquatic and terrestrial habitats that are necessary to support populations of crawfish frogs and spring peepers, both SINC species in Kansas, as well as American bullfrogs and gray treefrogs (Rohweder, 2015). Many anuran species were nearly ubiquitous in this area, suggesting that this altered landscape provides the appropriate habitats to support common anuran species.

Survey locations were split between the Spring River sub-basin and the Neosho River basin. Although there were no apparent defining landscape features that separated these watersheds, both SINC species were more likely to occupy the Spring River sub-basin. Spring peepers and crawfish frogs may have been found throughout this watershed because of the proximity to source populations to the east in Missouri, or because the region provided sufficient habitats allowing these species to be supported at the western edge of their distributional range.

Anthropogenic changes to this landscape have created a mosaic of interspersed cropland, forest, water, grassland, and impervious surfaces. In much of this area, cropland was one of the few places that still had deep topsoils, which crawfish frogs have been known to rely upon (Busby & Brecheisen, 1997). The mined lands were heavily disturbed with rocky tillage and minimal to no topsoil, which may have limited crawfish frogs' use or association with other habitats on the mined lands areas. Spring peepers had a strong negative association with cropland, likely because of their breeding habitat preferences of tree cover and temporary pools, which crop fields often cannot provide (Collins & Fahrig, 2017; Swanson et al., 2019).

The dramatic changes in soil structure associated with strip mining, and subsequent successional stages, have created waterbodies with varying hydroperiods and habitats. These habitats may not have existed in the absence of surface mining operations (Lannoo et al., 2009; Lannoo et al., 2014). The large open bodies of water, like the more permanent strip pits on mined lands or agricultural ponds, provided more American bullfrog breeding habitats, likely resulting in their strong association with the amount of open water on the landscape (Koumaris & Fahrig, 2016).

Spring peepers and gray treefrogs have been found to have variable breeding habitats, potentially making the amount of open water or wetlands less important at landscape scale relative to local scales, but allowed for these species to breed in the study area (Babbitt et al., 2003). Even though gray treefrog had percent wetland in the top model, this variable was not meaningful and did not explain their occupancy in this area. Cope's gray and gray treefrogs are notoriously difficult to distinguish by ear, especially as they often occupy similar habitats (Gerhardt, 2005). The inclusion of observer as a detection covariate suggests that there may be uncertainty of detection due to observer and competing calls, resulting in the uncertainty between specific landscape variables like percent of wetlands and gray treefrog occupancy.

Urbanization has been shown to negatively affect amphibian populations because of reduced aquatic and terrestrial habitat quality and availability (Rubbo & Kiesecker, 2005). In this study, the built environment included all impervious surfaces, primarily roads and urban development in the area. Spring peepers had a strong negative association with the built environment. Other studies have found that urbanization most often impacts amphibian species that require shallower, fishless ponds for breeding habitat, such as spring peepers (Hamer & Parris, 2011; Rubbo & Kiesecker, 2005). Species like American bullfrogs may, however, be more suited for urbanization, as urban ponds often provide the habitat needed for aquatic species that prefer permanent hydroperiods (Sauer et al., 2022). The negative association between spring peepers and the built environment was also likely driven by the limited dispersal capabilities within an urban environment, due to the increased amount of roads (Eigenbrod et al., 2008; Pillsbury & Miller, 2008). Although urbanization increases the noise and light at night which might decrease the detection of anuran species, urbanization has not been shown to decrease the occupancy of the area. Therefore association of spring peepers with the built environment is likely tied to availability of breeding habitats within urban areas (Cronin et al., 2022). Although this region's urban areas had relatively low population density (i.e., < 39,110 POP), it was likely impacting spring peepers in a similar way to larger population centers resulting in the strong negative association with the built landscape cover.

The native terrestrial habitats, like forest and grassland cover, did not have strong relationships to anuran occupancy in this study. Although gray treefrog had grassland in the top model, grassland cover showed a weak relationship to gray treefrog occupancy. Even so, the lack of support in our models for forest and grassland land cover does not indicate the lack of importance of these habitats on the landscape, as many studies with fewer anthropogenic changes to the landscape show relationships between species occupancy and forest and grassland cover. Forest is largely considered important for spring peepers and gray treefrogs, as this land cover type was often the preferred breeding habitat for these species (Collins & Fahrig, 2017; Eigenbrod et al., 2008; Knutson et al., 1999; Simpson et al., 2021). This pattern can be seen in the predictive plot for spring peepers; the predicted occupancy is highest on forested areas (Fig. 1.6A-B). As for grasslands, we classified hay fields and pastures as grassland land cover, which historically would have been tallgrass prairie in southeast Kansas. Anuran populations have been positively associated or have a neutral association with livestock and pasture land cover likely due to the lower intensity agricultural practices like pasture rotations or no-till row crops, which could be the driving force behind the predicted occupancy of crawfish frogs being in the cropland and grassland patches (Fig. 1.6; Howell et al., 2019; Koumaris & Fahrig, 2016).

Continued research is needed on anthropogenically altered landscapes to understand to a fuller extent how the landscape composition is influencing anuran populations, as some of the species' results had high levels of uncertainty. Our surveys were based on the MLWA to study the impacts of remnant strip mined areas, but most of this region has been affected by mining. Therefore, all land cover types are impacted. However, the addition of call sites not directly related to the MLWA would provide a clearer picture of how historic mining in the region influenced anuran occupancy, even for the species that were considered ubiquitous in this area. Additionally, modeling various landscape metrics like mean patch size, may provide a deeper understanding how the landscape mosaic is influencing anuran occupancy. The use of acoustic detectors could allow for a more accurate representation of gray treefrog occupancy on the landscape and provide information about differences in habitat preference between Cope’s gray and gray treefrogs.

In addition, focused research should target the SINC species in the area to further understand their relationship to landscape composition and land use. Spring peepers appeared on this landscape after most of the mining activity had concluded, with the first report of individuals in Cherokee Co. in 1951 (Rundquist, 1977) and in Crawford Co. in 2000 (Collins, 2001), suggesting that the land cover changes since mining have provided appropriate habitat for them to colonize the area. The mined lands continue to the west of spring peeper’s current range limit; thus, research could address the potential for spring peepers to extend their range. Examining the underlining causes for the association of crawfish frogs and croplands such as the amount of topsoil or connectivity to breeding ponds, would allow for a better understanding of habitat use in this anthropogenic landscape to provide support for conservation efforts.

CONCLUSION

Anthropogenetic changes to a landscape impact anuran occupancy in a variety of different ways. Even so, the variation and diversity in habitat types resulting from these changes may provide sufficient habitats to support anuran populations and communities. Due to the unique land use and mining history of this region, the availability of habitats such as forests, grasslands, open water, and wetlands, supports a variety of anuran populations, including SINC species. The management of aquatic and terrestrial habitats across all anthropogenetic landcover types will support current and future anuran population.

CHAPTER II

POST-MINED WETLANDS PROVIDE BREEDING HABITAT FOR AMPHIBIANS

ABSTRACT

Wetlands are complex, threatened ecosystems that have frequently become degraded over time. Post-mined landscapes can provide an increased number of wetlands, but little is known about the health of these wetlands on mined sites that have only been altered by vegetative succession, i.e., they have never been deliberately reclaimed. Amphibian persistence in wetlands in heavily disturbed ecosystems can help to determine the quality of habitat for amphibians and other wetland dependent species. This study aimed to describe the wetland characteristics that influence amphibian community composition and occupancy of individual species. Single species occupancy models were used to determine the wetland characteristics that influenced larval presence of five common species, including American bullfrog, (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), gray treefrog species complex (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*). The response of the amphibian community (i.e., richness, diversity, composition) to wetland features was examined through linear models and non-metric multidimensional scaling (NMDS). Occupancy for each species varied, but the presence of predatory fish, hydroperiod, and emergent vegetation cover were the most influential predictors of occupancy. Amphibian richness and diversity were influenced by the water conductivity level, the presence of predatory fish, hydroperiod, and emergent vegetation cover within the wetland. The NMDS showed that amphibian community composition was similar among wetlands regardless of the mining history or management. While species' occupancy patterns varied, the wetlands across the post-mined landscape provided sufficient habitat to support a diverse amphibian community. Increasing the variation in wetlands through protection, reclamation, and management could allow these amphibians and other wetland-dependent species to persist on the landscape.

INTRODUCTION

Wetlands are considered a threatened ecosystem globally, with 35% of wetlands lost since 1970 and even more degraded due to human disturbances, including changes in agriculture, urbanization, and surface mining (Dahl, 1990; Ramsar Convention on Wetlands, 2018). The degradation of wetlands across North America impacts the majority of amphibians that are dependent on wetland habitats (Church et al., 2008). Additionally, amphibians provide critical functions to ecosystems, such as the efficient transfer of biomass and nutrients between habitats (Burton & Likens, 1975; Hopkins, 2007; Semlitsch et al., 2014). Amphibians can also be used as water quality indicators due to their semipermeable skin; thus, they can provide habitat quality assessments for entire vertebrate communities (Boyer & Grue, 1995; Pollet & Bendell-Young, 2000). All of these features make amphibians useful in examining wetland health after disturbances such as surface mining.

Surface mining alters landscapes, destroys habitat, and disrupts ecosystem function by removing the top layer of earth to access mineral seams. In the decades following the Surface Mining Control and Reclamation Act (1997), mining companies have been required to reclaim disturbed mined areas with native habitats, with the goal of creating wetland habitats in and around the strip pits or deep rectangular pits left after mining activity. Multiple studies have assessed amphibian communities that use reclaimed wetlands on surface mined lands and found that natural and reclaimed wetlands had similar amphibian communities (Lannoo et al., 2014; Pollet & Bendell-Young, 2000; Sasaki et al., 2015; Stiles et al., 2017). This suggests that reclamation has created habitats that function similarly to natural wetland systems. However, most of previous studies focused on mined lands that were reclaimed in the years immediately following mining operations. Yet a large portion of surface mined lands were never reclaimed or were not reclaimed for decades after mining ended, resulting in vegetative succession.

Wetlands are complex ecosystems that have many biotic and abiotic factors that may influence the quality of breeding habitat for amphibians, like water quality, hydroperiod, vegetation, and predators. Water quality variables, such as water temperature, conductivity, dissolved oxygen (DO), and pH, have affected amphibian communities (Chambers, 2011; Karraker et al., 2008). For example, species richness responded positively to DO in an urban area, while pH can have either positive or negative effects, depending on the context (Brodman et al., 2003; Calderon et al., 2019; Camacho-Rozo & Urbina-Cardona, 2021). Conductivity is an important factor in heavily mined areas, as mining may introduce heavy metals and salts that increases conductivity, which can decrease the survival of amphibians (Chambers, 2011). Hydrologic conditions may be the most influential factor when creating wetlands, particularly the variety of hydroperiods that can support the large breeding populations of some amphibian species

(Brodman et al., 2003; Collinge et al., 2013; Nagel et al., 2021). Hydroperiod often drives other factors that influence breeding success, like the vegetation density and presence of predatory fish (Amburgey et al., 2014; Babbitt et al., 2003; Brodman, 2008). Aquatic vegetation can positively influence amphibian communities through the addition of microhabitats (Burne & Griffin, 2005; Hamer & Parris, 2011). Amphibian communities may also be negatively associated with the size class and density of predatory fish due to the increased predation rate of amphibian egg masses and larvae (Hartel et al., 2007; Kloskowski, 2009).

The present study aimed to identify wetland characteristics that affect the occupancy of five common amphibian species, which can be used as wetland indicators for the monitoring and management of the amphibian community. In addition, this study aimed to understand the biotic and abiotic characteristics of wetlands that influence amphibian communities (i.e., diversity and structure) at wetlands with a variety of mining histories and management activities. Information about the amphibian use of wetlands on previously mined lands can be used to determine conservation value and guide management practices of these disturbed systems to promote an ecosystem that supports a wide variety of biota.

METHODS

Study Area

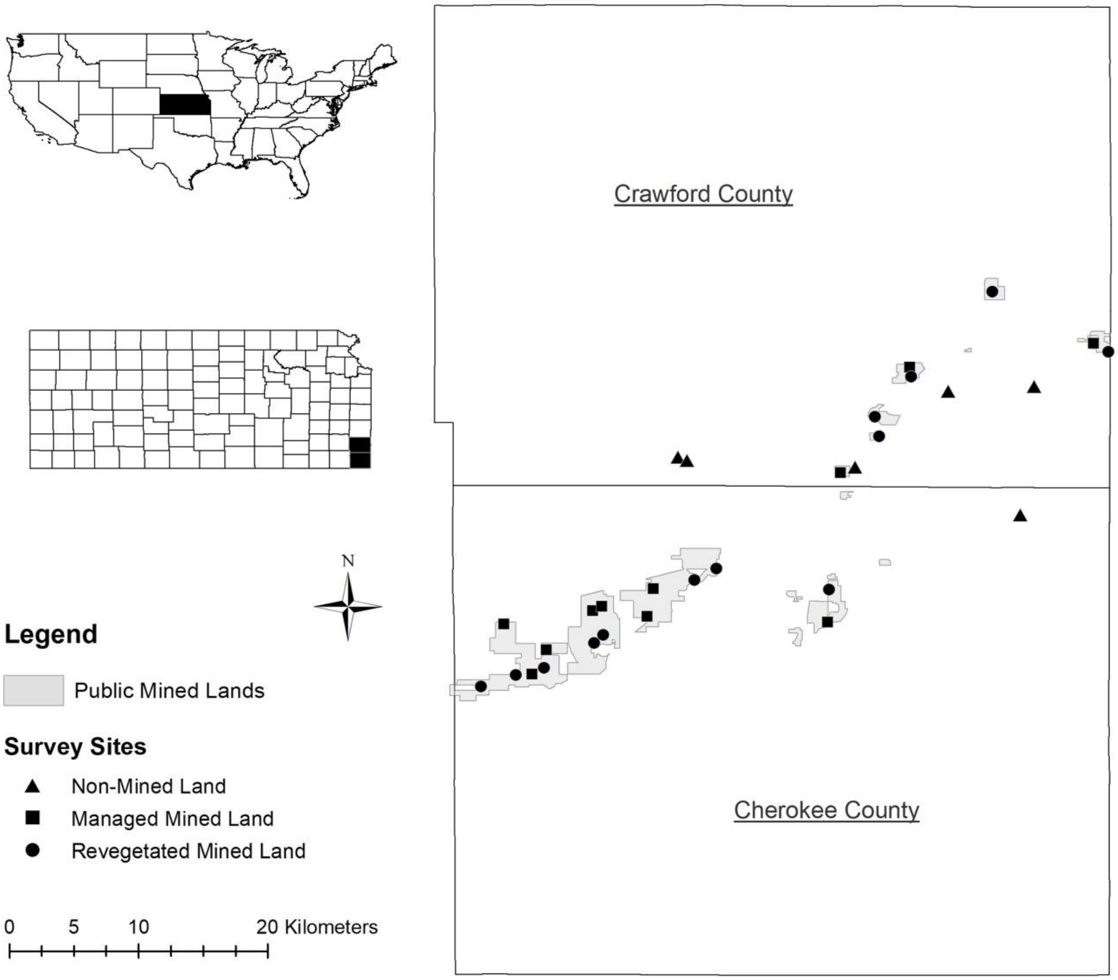

Figure 2.1. Map of the survey area with sites indicated by the mining and reclamation status of each wetland in southeast Kansas. Public mined lands in the area are shaded, including Mined Land Wildlife Area and Southeast Kansas Biological Station.

We sampled an area spanning southern Crawford Co. and northern Cherokee Co. in southeast Kansas. These counties are part of the Cherokee Lowlands physiographic region, which is characterized by rolling plains that have patches of forest along streams and on the abandoned mining areas (Fig. 2.1; Kansas Geological Survey 1999). This region was mined for coal and other metals from the 1850s to the 1980s, with most areas left unreclaimed, to be naturally revegetated (Bailey & Hooey, 2017; Kansas Historical Society, 2013). The Kansas Department of Wildlife and Parks (KDWP) has been working to reclaim 14,500 acres of historic strip-mined areas, which are collectively known as the Mined Land Wildlife Area (MLWA; KDPW, 2018). The KDWP has already converted some of this land into grasslands and marshes to help improve habitat quality for wildlife, such as waterfowl and Northern Bobwhite (*Colinus virginianus*). We surveyed wetlands within the MLWAs, the Southeast Kansas Biological Station, Buche Wildlife Area, and private properties, which included revegetated, managed, and non-mined sites.

Survey Methods

We chose 31 wetlands to survey across the study area that had varied mining histories and management activities (Fig. 2.1, Appendix VII). For the purpose of this paper, wetlands were categorized as revegetated (i.e., shallower [< 5m deep] wetlands on mined lands that have not been manually altered since mining), managed (i.e., wetlands on mined lands that were created or maintained by vegetation control and water-level manipulation), or non-mined (i.e., wetlands with no mining history whether actively managed or not). Of the selected wetlands, 13 were revegetated, 11 managed, and seven non-mined. Revegetated wetlands were chosen based on the accessibility of the waterline (i.e., low vegetation density and shallow slope). Most of the managed wetlands had been modified to support waterfowl hunting and were considered marshes by KDWP. Sites were located at least 300 m from each other and were scattered across the extent of the survey area (Fig. 2.1, Appendix VII). Two different wetlands at the Buche Wildlife Area were surveyed even though they were within 300 m of each other, but each was only surveyed in a single year.

To sample the amphibian community across varying breeding periods, we used minnow trap and dipnetting surveys for two consecutive days during three different survey windows defined as mid-March to mid-April (Early Spring), May (Spring), and June (Summer). We evenly spaced sampling locations along the shore from a random center point in either direction. We scaled distance between sampling locations by wetland size, ranging from 5 m between samples (wetlands < 0.05 ha), 10 m (wetlands between 0.05 ha to 0.35 ha), and 20 m (wetlands > 0.35 ha). Sampling locations alternated between minnow trap (four locations) and dipnetting (four locations) for a total of eight sampling locations per wetland. For each survey window, we adjusted the sampling locations within sites to account for seasonal fluctuations in the water line. Within each survey window, traps and dipnet locations were the same for both consecutive days.

We modified minnow traps (Gee’s Galvanized Wire Minnow Trap) with window screening and baited each trap for a random trap night per survey window with a green glow sticks to increase catch rate and potentially attract eastern newts (*Notophthalmus viridescens*), a species in need of conservation (SINC) in Kansas (Bennett et al., 2012; Grayson & Roe, 2007; Swartz & Miller, 2018). We placed traps at varying distances from shore to have the funnel entrance at least half covered with water while also having a portion of the trap exposed to the air for air-breathing species. At each dipnetting location, we extended the dipnet approximately 1 m from the waterline into the wetland and quickly pulled toward the shore along the bottom following a zig-zag motion (Babbitt et al., 2003). We conducted both surveys within a 24-hr period. We identified all invertebrates and vertebrates in samples based on field markings and recorded the total number captured for each trap and dipnet location before organisms were returned to the wetland. When species of fish were captured, we identified them to species level. Amphibians were identified based on field marks. Therefore, we couldn't distinguish among Cope's gray treefrog (*Hyla chrysoscelis*) and gray treefrog (*Hyla versicolor*) larvae and will hereafter refer to the gray treefrog complex as *Hyla* spp.

Wetland Characteristics

We recorded the area of the wetland with a Garmin eTrex 10 GPS unit. The surveyor walked the perimeter of the wetland and we calculated the wetland area based on the standing water line in Google Earth Pro (*Google Earth Pro*, 2022). We recorded the area with this method during each survey period unless the water level was not observably different. We considered the change in area across the three sample periods to be a proxy for hydroperiod. We sampled water quality once during each survey window. Water quality sampling included pH (HI 9812-5 Portable Meter), conductivity (µs/cm; HI 9812-5 Portable Meter; Babbitt et al., 2003), water temperature (ºC; YSI ProODO), and dissolved oxygen (DO; mg/L; YSI ProODO). During 2022 we replaced the water temperature and dissolved oxygen meter with an ExStik DO600 (Extech Instruments). Conductivity was later grouped into three categories (low < 500 µs/cm, medium = 500–1499 µs/cm, and high ≥1500 µs/cm), as some wetlands exceeded the range of the meter.

During each water quality sample, we recorded measurements from three random dipnet or trap locations chosen prior to sampling day, and all samples were taken approximately 1 m from the waterline (Babbitt et al., 2003). We visually estimated the percent cover of emergent vegetation within the wetland (Burne & Griffin, 2005). We also recorded the presence of predatory fish species, such as bass (*Micropterus* spp.), sunfish (*Lepomis* spp.), and gar (*Lepisosteus* sp.). The presence of these fish species was categorized at never, sometimes, or always based on captures through dipnetting and traps, and opportunistic sightings. The category of "sometimes" refers to sites that had a change in the presence of predatory fish between survey seasons, primarily due to flooding events connecting the wetland with a fish source, such as a nearby streams or other strip pit wetlands.

Data Analysis

We used an information theoretic approach to analyze the effects of wetland characteristics on individual species occupancy and wetland communities based on larval captures within each year (Burnham & Anderson, 2004). Before fitting models, we tested covariates for multicollinearity and excluded variables with $r > 0.7$ from the same models. We used Akaike's information criterion corrected for small sample size (AIC_c) to determine what models were supported by the data ($\Delta AIC_c < 2$).

We used the package "unmarked" in Program R (version 1.3.1073) to fit single-season occupancy models for five of the most common species found: American bullfrog, (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), *Hyla* spp., and southern leopard frog (*Lithobates sphenocephalus*). We fitted models in a stepwise process starting with detection probability using detection covariates. We used an additive approach to determine which covariates influence detection probability (Table 2.1). Detection covariates from the top models for detection were then included in all models for occupancy. We created models that estimated the probability of occupancy using the occupancy covariates (Table 2.1). We used an additive approach to determine occupancy covariates that were most influential based on AIC_c and model weights. We examined the model with the greatest number of parameters for the goodness of fit (MacKenzie & Bailey, 2004). If the model was overdispersed, we used $QAIC_c$ to compare the candidate model set.

We used the package “vegan” to analyze the larval amphibian community (Oksanen et al., 2022). We calculated Choa1 richness and Shannon diversity for each year at each site to describe the larval community diversity. We used linear models to examine the influence of wetland characteristics on the amphibian community (Table 2.1). We used an additive approach to determine the wetland characteristic variables that influenced richness and diversity. To compare the similarity of amphibian community structure between the three wetland types, we performed a non-metric multidimensional scaling (NMDS) ordination. Amphibian captures were included in a site × species matrix. We calculated the NMDS on Bray-Curtis distance matrices derived from a Wisconsin square root transformed capture numbers. We evaluated the stress to decide the number of ordination dimensions.

Table 2.1. Wetland characteristics included as covariates in liner models of amphibian species richness and diversity in southeast Kansas during 2021 and 2022. Superscripts indicate the use of the parameter for single species occupancy models for American bullfrog (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), Hyla spp. (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*). Temperature and day were only included in single species occupancy models. Parameters were averaged for each site within a single year of surveys.

Model Parameter	Description
day†	Ordinal date of survey
year†*	Year the survey took place (2021, 2022)
type†*	Classification of the wetland based on the history of the site (managed, non-mined, revegetated)
temp†	Average water temperature (°C)
pH*	Average pH of the wetland
DO*	Average dissolved oxygen of the wetland (mg/L)
cond*	Conductivity level based on average readings: low <500 µs/cm, medium = 500 – 1499 µs/cm, and high ≥1500 µs/cm
area*	Average area of the site (ha)
hydro*	Percent change in wetland area over the year as a relative proxy for hydroperiod
emveg*	Average percent cover of emergent vegetation in the wetland (%)
fish*	Presence of predatory fish (bass and sunfish; 0 = no presence, 1 = sometimes present, 2 = always present)

† Parameter used as a detection covariate for single species occupancy models
* Parameter used as an occupancy covariate for single species occupancy models

Results

We detected 10 amphibian species across the 31 sites in 2021 and 2022 (Appendix VIII). Most notably, three SINC species, crawfish frog (*Lithobates areolatus*), eastern newt, and spring peeper (*Pseudacris crucifer*), were captured at 9.7%, 6.4%, and 19.4% of the sites, respectively. Other species captured include American bullfrog (61.3% of sites), American toad (*Anaxyrus americanus*, 22.6%), Blanchard's cricket frog (90.3%), boreal chorus frog (41.9%), *Hyla* spp. (41.9%), smallmouth salamander (*Ambystoma texanum*, 22.6%), and southern leopard frog (83.9%). Richness estimation for sites in each year ranged from zero to nine, with a mean species richness of 3.37. Shannon Diversity for sites based on each year ranged from zero to 1.67 with a mean of 0.56.

Wetland occupancy varied by species. The best supported model for American bullfrog occupancy was the intercept-only for occupancy with site type affecting detection probability (Table 2.2; Appendix IX). The hydroperiod best explained occupancy for Blanchard's cricket frog (Table 2.2, Appendix X); the greater the change in wetland area over a year, the less likely cricket frogs were to occupy the wetland (Table 2.3, Fig. 2.2A). Boreal chorus frog occupancy was best explained by the average percent cover of emergent vegetation (Table 2.2, Appendix XI). Boreal chorus frog occupancy increased with more emergent vegetation within the wetland (Table 2.3, Fig. 2.2B). Although the presence of predatory fish was the best supported model for *Hyla* spp. (Table 2.4, Appendix XII), there was not a clear pattern for occupancy because of the large confidence intervals for predatory fish (Table 2.3, Fig. 2.2C). Lastly, the best supported model for southern leopard frog occupancy was the change in area of wetland over the survey period (Table 2.2, Appendix XIII). Southern leopard frog occupancy increased with a greater change in wetland area over the year (Table 2.3, Fig. 2.2D).

The best supported model for amphibian species richness included the presence of predatory fish, conductivity level, and percent cover of emergent vegetation within the wetland (Table 2.4, Appendix XIV). Richness was greatest in wetlands that sometimes had predatory fish, had low conductivity, and those with more emergent vegetation (Table 2.5, Fig. 2.3A–C). The best supported model for amphibian diversity included conductivity, presence of predatory fish, and the change in wetland area over the year (Table 2.4, Appendix XV). Amphibian diversity was the highest in wetlands with low conductivity, and that sometimes had predatory fish, and showed a negative relationship with percent change in wetland area within a year (Table 2.5, Fig. 2.3D – F).

The first two dimensions of the NMDS ordination had a goodness of fit of 0.17, suggesting there was a fair representation of dissimilarity between wetland in reduced dimensions, but some distances may be misleading. We did not find strong evidence that amphibian community structure differed among non-mined, managed, and revegetated sites; however, the non-mined and revegetated wetlands showed some differentiation in composition from each other (Fig. 2.4A). The wetland characteristics of predatory fish presence, DO, conductivity level, wetland type, percent change in wetland area, and percent emergent vegetation within the wetland were associated with the ordination scores between the sites (Fig 2.4B).

Table 2.2. Top-ranked ($\Delta QAIC_c$ <2) occupancy models estimating the probability that American bullfrog (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), *Hyla* spp. (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*) occupied wetland sites during 2021 and 2022 in southeast Kansas. Due to overdispersion, $QAIC_c$ was used for all species except boreal chorus frog, in which case AIC_c was used. Null models have also been included, along with each model's parameters (K) and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
American bullfrog			
p(type) ψ(.)	5	0	0.17
p(.) ψ(.)	3	0.58	0.13
p(type) ψ(pH)	6	1.09	0.10
p(type) ψ(hydro)	6	1.60	0.08
p(type) ψ(emveg)	6	1.64	0.08
p(type) ψ(area)	6	1.82	0.07
Blanchard's cricket frog			
p(day) ψ(hydro)	5	0	0.36
p(day) ψ(hydro + area)	6	1.29	0.19
p(day) ψ(.)	4	2.45	0.11
p(.) ψ(.)	3	21.61	0
Boreal chorus frog			
p(day) ψ(emveg)	4	0	0.46
p(day) ψ(emveg + DO)	5	1.40	0.23
p(day) ψ(.)	3	7.33	0.01
p(.) ψ(.)	2	24.47	0
Hyla spp.			
p(day) ψ(fish)	6	0	0.23
p(day) ψ(fish + cond)	8	1.02	0.14
p(day) ψ(fish + area)	7	1.69	0.10
p(day) ψ(fish + emveg)	7	1.95	0.09
p(day) ψ(.)	4	6.04	0.01
p(.) ψ(.)	3	11.95	0

Table 2.2. Continued.

Model	K	Δ $QAIC_c$	Model Weight
Southern leopard frog			
p(type + day) ψ(hydro)	7	0	0.22
p(type + day) ψ(.)	6	0.85	0.14
p(.) ψ(.)	3	12.48	0

Table 2.3. Estimates of each occupancy parameter with the top occupancy model for Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), *Hyla* spp. (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*), based on wetland surveys conducted in 2021 and 2021 in southeast Kansas. Beta estimates, standard errors (SE), and the lower and upper 95% confidence interval (CI) are included for each parameter.

	Parameter	Estimate	SE	Lower 95% CI	Upper 95% CI
Blanchard's cricket frog	intercept	2.99	1.23	0.96	5.01
	hydro	-3.39	1.67	-6.14	-0.64
Boreal chorus frog	intercept	-1.83	0.52	-2.69	-0.97
	emveg	0.05	0.02	0.02	0.07
Hyla spp.	intercept	0.21	0.38	-0.42	0.84
	sometimes fish	-0.61	0.70	-1.76	0.54
	always fish	-9.53	26.13	-52.51	33.44
Southern leopard frog	intercept	0.26	0.42	-0.43	0.96
	hydro	4.05	1.80	1.09	7.01

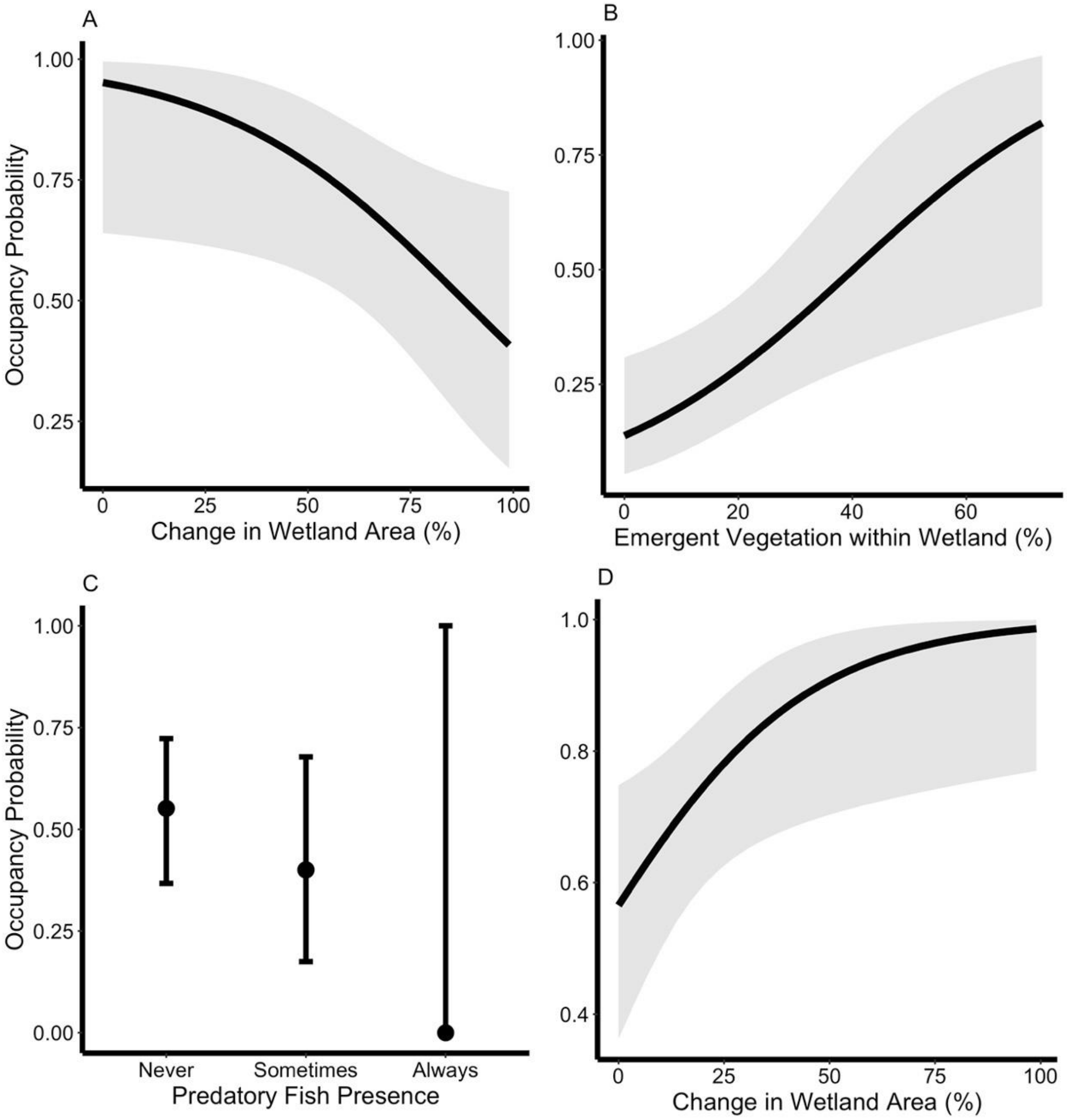

Figure 2.2. Predictive plots based on the top occupancy model for A) Blanchard's cricket frog (*Acris blanchardi*), B) boreal chorus frog (*Pseudacris maculata*), C) *Hyla* spp. (*Hyla chrysoscelis/versicolor*), and D) southern leopard frog (*Lithobates sphenocephalus*) in wetlands during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

Table 2.4. Top models (ΔAICc <2) of the effects of wetland characteristics on the amphibian species richness and Shannon diversity in wetlands across southeast Kansas during 2021 and 2022. Null models have also been included, along with each model's parameters (K) and weights. See Table 2.1 for variable definitions.

Response Variable	Model	K	ΔAICc	Weight
Richness	fish + cond + emveg	7	0	0.41
	fish + cond	6	0.69	0.29
	fish + cond + do	7	0.88	0.26
	null	2	26.85	0.00
Diversity	cond + fish + hydro	7	0	0.75
	null	2	13.23	0

Table 2.5. Estimated coefficients for the top model of the effects of wetland characteristics on the species richness and Shannon diversity index of amphibian species in wetlands across southeast Kansas during 2021 and 2022.

Response Variable	Parameter	Estimate	Standard Error	Lower 95% CI	Upper 95% CI
Richness	intercept	4.14	0.45	3.38	4.90
	sometimes fish	0.81	0.55	-0.11	1.74
	always fish	-1.82	0.52	-2.69	-0.94
	medium cond	-1.36	0.48	-2.16	-0.55
	high cond	-2.27	0.60	-3.28	-1.26
	emveg	0.02	0.01	0.00	0.04
Diversity	intercept	0.95	0.11	0.76	1.13
	medium cond	-0.27	0.12	-0.47	-0.08
	high cond	-0.43	0.15	-0.69	-0.17
	sometimes fish	0.14	0.13	-0.09	0.36
	always fish	-0.38	0.13	-0.59	-0.16
	hydro	-0.42	0.18	-0.72	-0.13

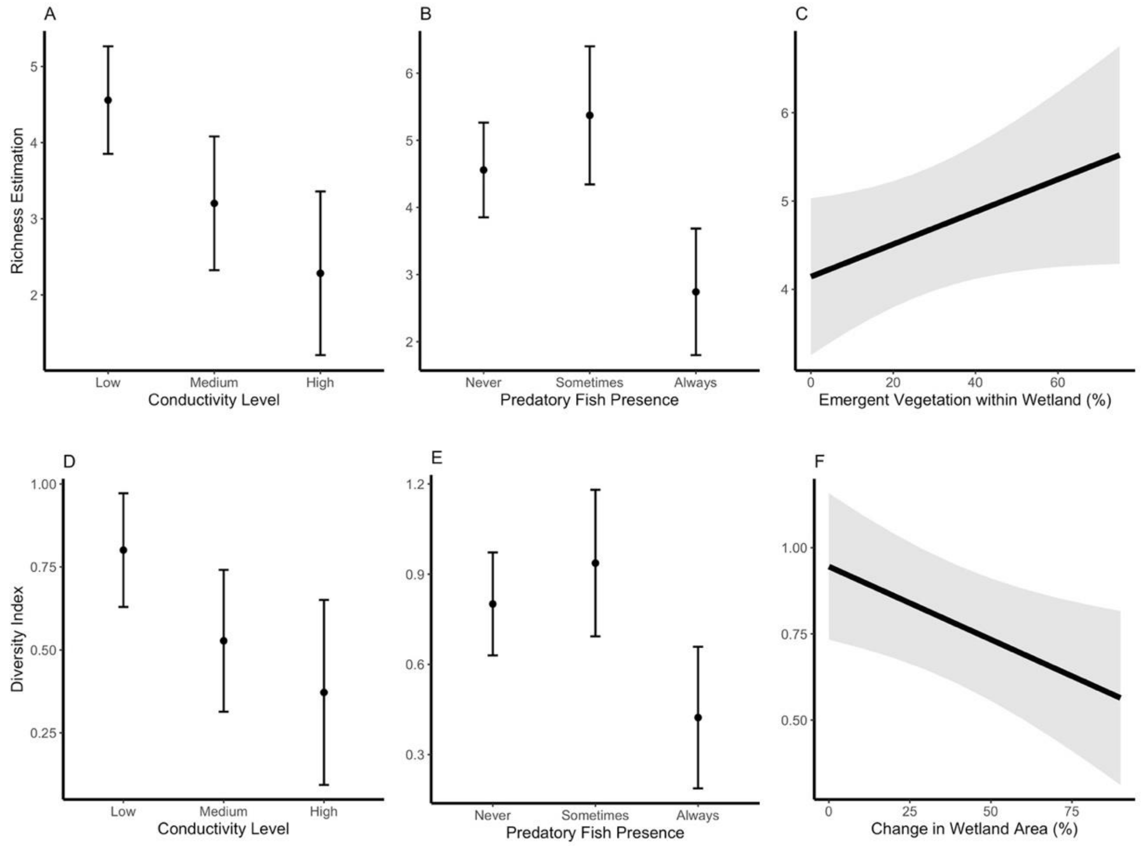

Figure 2.3. Wetland characteristics that influence the species richness and Shannon diversity of amphibian communities in southeast Kansas in 2021 and 2022. Species richness was affected by A) conductivity level, B) the presence of predatory fish, and C) percent of emergent vegetation within wetlands. Diversity was affected by D) the conductivity level, E) the presence of predatory fish, and F) the percent change in wetland area within a year. Fish presence was measured as the level of predator fish species (e.g., bass, sunfish, and gar) presence, where “sometimes” refers to the change in fish presence within a year. Conductivity level was measured as low (< 500 µs/cm), medium (500 – 1499 µs/cm), and high (≥ 1500 µs/cm).

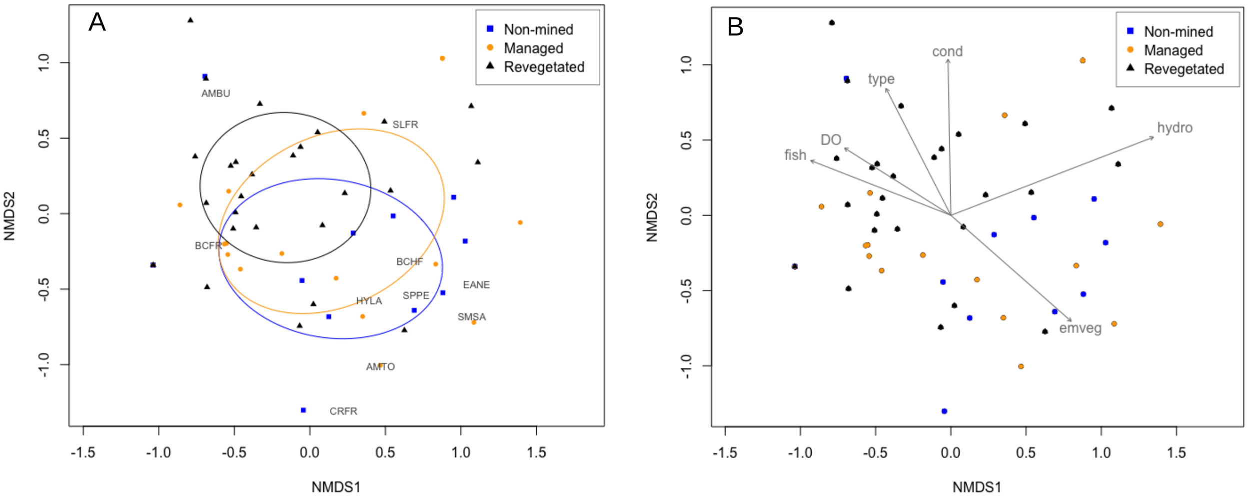

Figure 2.4. NMDS ordination plot showing amphibian community structure in southeast Kansas during 2021 and 2022. A) Wetland type was used to depict differences between communities, with the ellipses representing the standard deviations of site scores. Letters represent species codes: AMBU = American bullfrog, AMTO = American toad, BCFR = Blanchard's cricket frog, BCHF = boreal chorus frog, EANE = eastern newt, *Hyla* = *Hyla* spp., SMSA = smallmouth salamander, SLFR = southern leopard frog, and SPPE = spring peeper. B) Amphibian community structure associated with wetland characteristics (variables with $p < 0.05$). Arrows represent the direction and magnitude of the wetland characteristics in relation to the wetland communities. Points represent sites with the color and shape indicating the wetland type of each site.

Discussion

The post-mined landscape provided larval habitats for individual species and for the entire amphibian community. Occupancy of individual species varied, but generally was associated with the absence of predatory fish, hydroperiod and percent cover of emergent vegetation. Richness and diversity of the larval amphibian community were associated with conductivity levels, predatory fish presence, percent change in wetland area, and percent of emergent vegetation within the wetland.

Conductivity and other water quality metrics are generally considered important for the survival of amphibians due to their semi-permeable skin that can absorb pollutants and other impurities that can alter their behavior and development (Chambers, 2011; Karraker et al., 2008; Pollet & Bendell-Young, 2000). We found that lower conductivity levels were associated with higher species richness and diversity. High conductivity levels can disrupt larval behavior and decrease survival rates (Chambers, 2011). Conductivity measurements in waterways are often associated with the increased runoff of road salt, which can decrease the number of egg masses and larvae (Karraker et al., 2008). However, in our survey area the primary cause for higher conductivity was likely not from road salt, but instead from heavy metals from mining activities, as mining can release various heavy metals into the water (Evans et al., 2021). Although the amphibian communities did not vary between sites with different mining histories and management, sites with high conductivity were more often revegetated wetlands.

Risk of predation can also influence amphibian communities. For example, limiting fish, both predatory and non-predatory, or the absence of predatory fish, can increase amphibian species richness and occupancy (Boone et al., 2007; Hartel et al., 2007; Hecnar & M’Closkey, 1997). Particularly, the age and size structure of predatory fish were meaningful in describing the diversity of amphibian communities, with wetlands containing larger, older fish resulting in smaller amphibian populations and communities (Kloskowski, 2009). By breaking up the presence of predatory fish into three categories, we were able to examine the change in fish presence over the season and its influence on amphibian communities and *Hyla* spp. occupancy. The complete and partial absence of predatory fish increased the likelihood of *Hyla* spp. occupancy and increased richness and diversity. Often wetlands that sometimes had predatory fish were smaller and more likely to dry out completely each year. Thus, these wetlands only had the addition of predatory fish due to flooding events that connected smaller wetlands with larger, deeper wetlands. The temporary influx of predatory fish decreased the number of amphibians in the short term, but likely allowed for a quick return to high quality habitat for a large number of amphibian species due to the limited time that larger fish persisted within that wetland system (Kloskowski, 2009).

The hydroperiod of wetlands often indicates which species will breed in a wetland because other characteristics, like fish and vegetation, are often a result of the hydroperiod (Brodman, 2008). Wetlands with shorter hydroperiods are often associated with emergent vegetation and fewer fish because there are seasonally dry periods with little or no standing water, while longer hydroperiods are more likely to contain predatory fish and limit emergent vegetation. Although we did not directly measure the hydroperiod of each wetland, the percent change in wetland area over the survey period can be used as a proxy because the wetlands that dry early will likely stay dry and the wetlands with no size differences likely are wet year-round. We found that percent change in wetland area and emergent vegetation were informative for a number of our studied species and the overall amphibian community.

Blanchard's cricket frog occupancy was negatively associated with the percent change in wetland area between March and June. The relatively quick rate at which some wetlands may dry has been shown to decrease the survival rate of cricket frog larvae, which can ultimately decrease the occupancy in the wetlands that regularly dry quickly and early in the year (Gordon et al., 2016). On the other hand, southern leopard frog occupancy was positively associated with the percent change in wetland area. Southern leopard frogs breed early in the year, allowing for increased time to reach metamorphosis. The positive association with a shorter hydroperiod is likely the driving force for other wetland characteristics that supported the occupancy of other species, such as the absence of predatory fish and increased emergent vegetation.

Emergent vegetation influenced boreal chorus frog occupancy and amphibian species richness. Increased emergent vegetation within a wetland provides expanded microhabitats that can support an increased number of species (Burne & Griffin, 2005). Boreal chorus frogs breed in early spring when there is limited emergent vegetation, so emergent vegetation likely represents other wetland characteristics that chorus frogs prefer in their breeding habitat. More emergent vegetation within a wetland often indicates that the wetland is shallow, which can lead to periodic drying events that limit the number of predatory fish species.

Wetland characteristics can often easily be seen as correlated with one another, making it a challenge to determine which characteristic is driving others that may be influencing occupancy of individual species and the entire amphibian community. The manipulation of wetlands has various impacts on the amphibian community, which are often driven by how the characteristics of natural wetlands are represented in reclaimed or managed wetlands (Brown et al., 2012; McPherson et al., 2020; Shulse et al., 2010, 2012). The amphibian community was not delimited by the type of mining and management history that a wetland had undergone, but some individual species, like American bullfrog, Blanchard's cricket frog, and southern leopard frog, tended to occur specific sites that were not occupied by other species in any large numbers. This similarity in amphibian communities shows that the wetlands across the landscape provided the variation and habitat conditions needed to support the full community. Although this study primarily focused on the mined land wetlands, the sampling region is surrounded by farm ponds, which also can support breeding populations of amphibians (Swartz & Miller, 2021). The combination of mined land wetlands and pond wetlands on the landscape may provide the diversity and connectivity of wetland habitats necessary to promote highly diverse communities or sustained populations (Brodman, 2008; Gibbs, 2000).

Reclaimed mined lands are often used for recreational opportunities, like hunting and fishing. Common practices for the management of deep wetlands include stocking game fish like trout and bass, or seasonal draining to increase waterfowl habitat for the winter hunting season. These practices may be beneficial to amphibians by providing a variety of wetland conditions, including the absence of predatory fish, hydroperiod, and amount of emergent vegetation. However, these practices may also be detrimental to some individual species that have other requirements that are not being met by management for waterfowl or fish. For example, the eastern newt (Kansas SINC species) was found to be breeding in only two ponds, one of which was actively managed for waterfowl by drying and tilling the area by July (Buckardt et al., 2022). This early drying likely limits survival rates of larvae prior to and after metamorphosis, due to the decreased larval sizes (Werner, 1986).

Our study demonstrates that variability among wetlands across the landscape provided common and SINC amphibian species with diverse habitats, although there was some uncertainty in our results. Variation in water quality, fish presence, emergent vegetation, and hydroperiod at various wetlands allowed the larval amphibian community to persist in a post-mined landscape. The continued protection and reclamation of wetlands could mitigate amphibian population declines and support other vertebrate communities that use wetlands (Ramsar Convention on Wetlands, 2018). Variation in the timing of management practice for managed wetlands can help to mimic the natural variations that occur from year to year and therefore help promote a more diverse biotic community.

CHAPTER III

TO GLOW OR NOT TO GLOW: EFFECTIVENESS OF GLOW STICKS AND TRAP METHOD ON THE CAPTURE RATES OF LARVAL AMPHIBIANS

ABSTRACT

Monitoring larval amphibians can be an important part of assessing populations and wetland health. The preferred methods to capture larval amphibians often vary based on the research question and logistics. But the relative efficacy of differing survey methods for a given species is often unknown. We aimed to examine how the capture rates for all amphibian larvae and five focal larval species were affected by season and survey method (i.e., dipnetting, un-baited minnow traps, and baited minnow traps). We surveyed 28 wetlands for amphibian larvae from mid-March to the end of June during 2021 and 2022 in southeast Kansas. We surveyed each wetland three times each year with 4 dipnet and 4 minnow trap locations for a 48-hr period, resulting in a total of 681 dipnet and 664 trap sampling locations. Green glow sticks were randomly placed in traps for a 24-hr period during each survey event, resulting in 1327 trap nights. We used generalized linear mixed-effects models to determine the effects of time of year and survey method on capture rates of individual larval species and of all larval species. Capture rates for total amphibians, American bullfrog (*Lithobates catesbeianus*), *Hyla* spp., and southern leopard frog (*Lithobates sphenocephalus*) changed over the seasons, depending on capture method. Capture rates of American bullfrogs and Blanchard's cricket frogs (*Acris blanchardi*) changed throughout the year based on the presence of bait. Minnow traps baited with glowsticks increased the total number of amphibian larvae captured, but these effects varied for individual species. The choice of dipnets, baited traps and un-baited traps for sampling larval amphibians needs to be carefully considered but using both methods may provide a more complete understanding of the larval wetland community.

INTRODUCTION

Amphibians are considered indicators of wetland health because their physiology and life history are closely tied to the wetland conditions(Taylor et al., 2020). Thus, reliable sampling methodology is important for assessing both amphibian populations and for monitoring wetland dynamics over time. Methods of capturing aquatic larvae range from dipnetting and seining to various types of traps; each technique has its own benefits and drawbacks (Skelly & Richardson, 2009). Research question and logistics are often the deciding factors between these methods. The choice can become more difficult when the relative merits of each method are unknown. Thus, to ensure studies can fit within a project's limited time and resources, survey methods should be compared to maximize larval amphibian capture rates.

Common survey methods for aquatic amphibians are dipnetting and trapping. Dipnetting has the advantage when a study has limited resources, as there is minimal equipment costs and surveys at a single location can be completed in one day (Skelly & Richardson, 2009). On the other hand, trapping requires more time, equipment, and often additional labor because a single project may require multiple traps to be checked within 24 hrs (Skelly & Richardson, 2009). Dipnet surveys have the potential to disturb habitat by scrapping the bottom of the wetland, which can change the microhabitats that the egg masses and larvae need. Trapping minimizes these disturbances because the traps are placed on top of the substrate or within the water column (Richter, 1995). Additionally, the decision to trap over dipnet, or vice versa, may depend on the project's focal species. Elusive or nocturnal species such as the greater siren (*Siren lacertina*) and the two-toed amphiuma (*Amphiuma means*) may be more easily captured with traps (Denton & Richter, 2012; Johnson & Barichivich, 2004; Willson et al., 2011).

The use of bait within a trap may attract target species, increasing their capture rates. Recent studies have demonstrated increased trap capture rates with the use of glow sticks, particularly for salamanders like the eastern newt (*Notophthalmus viridescens*; Grayson & Roe, 2007) and tiger salamander (*Ambystoma tigrinum*; Liebgold & Carleton, 2020). These baited traps captured other amphibian species in addition to the target species; however, the efficacy of glowstick-baited traps for other amphibian larvae and adults has not been examined.

Our goal was to assess the effectiveness of dipnetting, minnow traps, and the use of glow sticks as bait for capturing larval amphibians. We compared catch per unit effort (CPUE) between a standardized dipnet method and a modified metal minnow trap, as well as the CPUE for glow stick baited and un-baited traps. We predicted that there would be an increased total CPUE for baited traps, but that CPUE would differ by species due to varying attraction to the green light. Documenting capture rates for alternate methods will allow researchers to more accurately assess larval amphibian communities.

METHODS

We sampled larval amphibians at 28 wetlands with known breeding populations across southeast Kansas. We surveyed for two consecutive days during three different survey windows defined as mid-March to mid-April (Early Spring), May (Spring), and June (Summer) during 2021 and 2022 to account for the variability of breeding times of anuran species in Kansas. We evenly spaced sampling locations along the shoreline starting from a random center point and placed sampling locations in both directions. We scaled distance between sampling locations by wetland size, ranging from 5 m between samples (wetlands < 0.05 ha), 10 m (wetlands between 0.05 ha to 0.35 ha), and 20 m (wetlands > 0.35 ha). We alternated between placing a minnow trap (four locations) and dipnetting (four locations) along the transects, for a total of eight sampling locations per wetland. For each survey window, the sampling locations within a site changed based on the fluctuating water line throughout the spring and summer. We also attempted to sample as much of the wetland edge as possible. This study design resulted in a total of 681 dipnet and 664 trap sampling locations in 2021 and 2022. The difference between the number of dipnet and trapping locations is due to lower water levels at some wetlands due to summer drying. There were1327 trap nights (24hr period) in 2021 and 2022.

We used modified minnow traps (Gee’s Galvanized Wire Minnow Trap) with window screening to ensure that smaller larvae could be captured (Skelly & Richardson, 2009; Swartz & Miller, 2018). We baited each trap with a green glow stick for one night per survey window with a green glow stick (Glow with Us 6” light sticks), resulting in most nights having only two traps baited at a single time. Traps were placed at varying distances from the shore to ensure that the funnel entrance was at least half covered with water, while also having a portion of the trap exposed to the air for air-breathing species.

We checked traps after ca 24 hrs, removed spent glow sticks, and baited the previously un-baited traps.

We used a D-frame dipnet to conduct dipnet surveys. At each dipnetting location, we extended the dipnet approximately 1 m from the waterline into the wetland and quickly pulled it toward the shore along the bottom following a zig-zag motion (Babbitt et al., 2003). We dipnetted at the same locations on the two consecutive days of the sampling period. We identified and counted all amphibian larvae in the traps and dipnets in the field prior to releasing them at their capture sites.

We used generalized linear mixed models with a Poisson error distribution using the lme4 package in R to compare the effects of survey method on CPUE. (Bates et al., 2022; R Core Team, 2020). We compared the total number of larvae captured (CPUE) in dipnets and traps for a 48-hr period during each season because sampling locations remained in the same during this time. Models included site and year as random effects to account for non-independence of samples from the same time periods and locations. Fixed effects included survey method, survey season, and their interaction. To examine the effect glowstick baited and un-baited traps on capture rates, we used the CPUE of a singular trap night (ca. 24-hr period) to account for non-independence of samples from the same wetland and year. Random effects included year and site and fixed effects included day or year, baited or un-baited, and their interaction.

We examined models for the total larval amphibian CPUE and for five focal species that were captured in high enough quantities for analysis, including American bullfrog (*Lithobates catesbeianus*), Blanchard’s cricket frog (*Acris blanchardi*), gray treefrog complex (*Hyla chrysoscelis/versicolor*; hereafter referred to as *Hyla* spp.), smallmouth salamander (*Ambystoma texanum*), and southern leopard frog (*Lithobates sphenocephalus*). Models for individual species were confined to only survey windows that represented when larvae could be captured based on local breeding phenology (Taggart, 2022). The number of sites included for each species was also limited to sites with breeding presence to prevent overinflating zero captures.

RESULTS

We captured 10 species of larval amphibians, including American bullfrog, American toad (*Anaxyrus americanus*), Blanchard's cricket frog, boreal chorus frog (*Pseudacris maculata*), crawfish frog (*Lithobates areolatus*), eastern newt, *Hyla* spp., smallmouth salamander, southern leopard frog, and spring peeper (*Pseudacris crucifer*). All species were captured using the various survey methods except the eastern newt larvae, which were captured by dipnet only.

Dipnet and trap CPUE for all amphibian larvae (n = 1345) was related to the interaction effect between season and survey method (Fig. 3.1). Traps captured more individuals than dipnets in the spring and summer while dipnet captures stayed consistent over the seasons (Fig. 3.1). American bullfrog larvae (n = 552) capture rates were higher in traps in the early spring, but this species had a similar CPUE between survey methods during the spring and summer (Fig. 3.1). CPUE for Blanchard's cricket frog (n = 720) was best explained by season and capture method, with more captures in the summer and with a dipnet (Fig. 3.1). *Hyla* spp.'s (n = 384) CPUE also increased in the summer and with a dipnet (Fig. 3.1). The CPUE for southern leopard frogs (n = 1177) was related to the interaction between survey method and season, with more captures in traps later in the year, but similar CPUE for dipnet throughout the year (Fig. 3.1). Smallmouth salamander (n = 208) capture rates increased using traps and during the spring (Fig. 3.1).

The total number of amphibians captured by traps (n= 1327) increased with day of year and increased with the use of glow stick bait (Fig. 3.2). *Hyla* spp. (n= 367) capture rates increased with the day of the year, but did not change with the bait presence (Fig. 3.2). Smallmouth salamander (n = 159) capture rates decreased with the day of year and there was no meaningful difference in capture rates with the use of bait (Fig. 3.2). Southern leopard frog (n = 1159) capture rates increased with the day of year, and with baited traps (Fig. 3.2). American bullfrog (n = 648) capture rates were best explained by an interaction between baiting and day of year. Capture rates for American bullfrog decreased in baited traps over the year, while capture rates remained constant for un-baited traps (Fig. 3.2). Blanchard's cricket frog (n = 743) captures also had an interaction effect; as the year progressed, capture rate slightly increased for traps that were baited (Fig. 3.2).

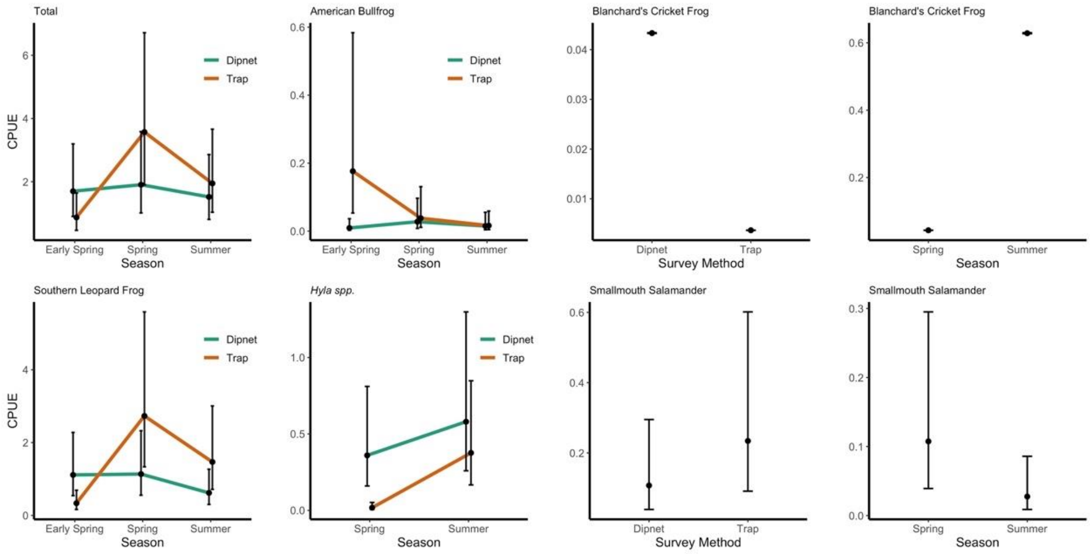

Figure 3.1. The effects of survey method and season (early spring, spring, and summer) on the capture per unit effort (CPUE) of larval amphibians in wetlands in southeast Kansas during 2021 and 2022. Capture rates for total amphibians, American bullfrog (*Lithobates catesbeianus*), *Hyla* spp., and southern leopard frog (*Lithobates sphenocephalus*) differed by survey method and season. Season and survey method affected CPUE for Blanchard's cricket frog (*Acris blanchardi*) and smallmouth salamander (*Ambystoma texanum*). Blanchard's cricket frog, *Hyla* spp., and smallmouth salamanders only were examined for spring and summer, as larvae were only found during those seasons. Error bars and shading indicate 95% confident intervals.

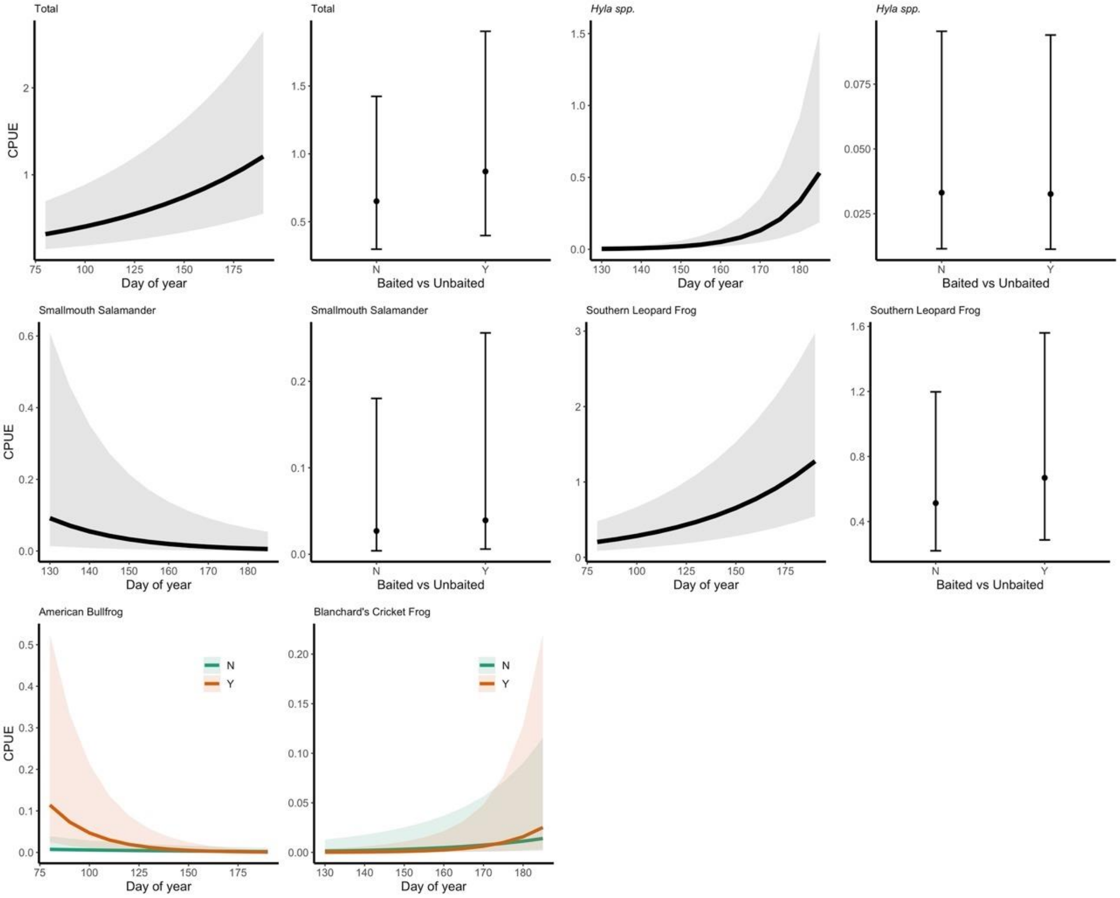

Figure 3.2. The effects of baiting minnow traps with glow sticks and season (early spring, spring, and summer) on the capture per unit effort of larval amphibians in wetland in southeast Kansas during 2021 and 2022. Capture rates for total amphibians, *Hyla* spp., smallmouth salamander (*Ambystoma texanum*) and southern leopard frog (*Lithobates sphenocephalus*) were affected by day of year and bait presences. American bullfrog (*Lithobates catesbeianus*) and Blanchard's cricket frog (*Acris blanchardi*) capture rates changed throughout the year based on bait presence. Error bars and shading indicate 95% confident intervals.

DISCUSSION

While the effects of capture method and baiting varied by species, minnow traps baited with glowsticks increased the total number of amphibian larvae captured. The optimal method to use will depend on the season and target species. To maximize the capture rates for larvae, the use of traps during the time of year that a wetland has the highest number of larvae present.

Although traps produced higher capture rates for most species because of their passive approach to capturing individuals, *Hyla* spp. and Blanchard's cricket frogs were primarily captured by dipnet. These species have similar breeding times, restricting their potential sampling to only the spring and summer, and thus limiting the size of larvae available when wetlands were surveyed. While we modified our minnow traps to decrease mesh size, smaller bodied larvae were not captured as often in traps compared to dipnets. Thus, our findings may have been skewed towards species with larger larvae such as American bullfrog and southern leopard frog, causing the differences in CPUE between survey method. Varying capture rates for *Hyla* spp. between survey methods have been reported elsewhere, suggesting that other factors are at play such as time of year and wetland characteristics (Denton & Richter, 2012). The change in the effectiveness of either survey method through the year is likely linked to breeding cycle, where the highest capture rates for larvae are the season directly after the primary calling period of the adults.

The presence of glow sticks in the minnow traps reflected the difference between dipnetting and trapping but showed an overall smaller effect. The use of glowsticks increased the capture rates of eastern newts and American bullfrogs, suggesting that light as a bait source is beneficial to capture rates for a least some species (Grayson & Roe, 2007; Liebgold & Carleton, 2020). The use of light bait can be especially important when studying species of conservation concern that are rare on the landscape. For example, eastern newts are a state threatened species in Kansas and a county record was discovered with the use of a glowstick-baited trap (Buckardt et al., 2021; Rohweder, 2015). Although this record was an adult newt, the use of glowstick-baited traps for this species of conservation concern helped to find a population that may not have otherwise been detected. Even so, the use of a glow stick was not a universal attractant; we did not detect differences in CPUE between baited and un-baited traps for *Hyla* spp., smallmouth salamander, and Blanchard's cricket frog. Future research could lead to improved techniques for effectively sampling larvae of these species.

Changes to the effectiveness of glowsticks over the year are likely influenced by multiple factors. The length of the chemical reaction in glow sticks can change based on the temperature, with a longer glow time in colder temperatures. The longer light source in the trap may increase the number of individual larvae captured. Wetland characteristics such as vegetation may limit the visibility of the light source, as more vegetation may block the light and decrease the chances of it being seen by individuals that are farther away.

Although our study included two years of data across multiple sites, there are limitations that should be considered. The wetlands in this study were primarily pond-like habitats with limited emergent vegetation and relatively long hydroperiods. These features may increase the likelihood that species such as spring peeper or crawfish frog may be found (Babbitt et al., 2003). Additionally, we designed this survey (i.e., number and placement of traps) with the goal of examining the entire larval amphibian community. Although each dipnet and trap location was considered independent, adjacent locations may have been influenced by similar factors such as presence of a single artificial light source. Since we were examining the larval communities, we did not test the use of light bait on capture rates of adult amphibians, which likely differs due to their mobility and diet changes after metamorphous.

The use of dipnets, baited traps and un-baited traps for sampling larval amphibians should be carefully considered, as capture rates of individual species may differ. Using both methods when examining the entire community instead of a single species may provide a more complete understanding of a wetland community. The choice of methodology should be decided by the research question, logistics, and other factors like wetland habitat characteristics.

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APPENDIX

Appendix I. Location name, survey group, and coordinates for each call survey site. Asterisks indicate private property. Coordinates for sites on private property have been removed to protect the landowner's privacy.

Location	Latitude	Longitude
Buche Wildlife Area	37.316974	-94.682287
Ford N*	-	-
Ford S*	-	-
MLWA 10	37.266704	-94.809560
MLWA 11	37.265700	-94.837879
MLWA 12N	37.258830	-94.815712
MLWA 12W	37.252120	-94.823983
MLWA 13	37.251744	-94.800832
MLWA 14	37.244293	-94.814228
MLWA 16	37.236934	-94.832718
MLWA 17S	37.287281	-94.894402
MLWA 17W	37.294005	-94.904708
MLWA 18E	37.274680	-94.908684
MLWA 18N	37.278798	-94.922917
MLWA 18S	37.266982	-94.914834
MLWA 19	37.278018	-94.895768
MLWA 1E	37.477094	-94.692814
MLWA 1N	37.482111	-94.702619
MLWA 1S	37.470528	-94.702748
MLWA 21E	37.246787	-94.960168
MLWA 21S	37.237713	-94.961258
MLWA 21W	37.245497	-94.976008
MLWA 22E	37.231035	-94.983160
MLWA 22S	37.223694	-94.990934
MLWA 23	37.236269	-94.973374
MLWA 24E	37.208764	-95.001307
MLWA 24W	37.212982	-95.011926
MLWA 25	37.193670	-95.059222
MLWA 26	37.332893	-94.800483
MLWA 27	37.202004	-95.050163
MLWA 28	37.202911	-95.031941
MLWA 29	37.201895	-95.013651
MLWA 3	37.443976	-94.617400

MLWA 30	37.208272	-95.022606
MLWA 32	37.208684	-94.977684
MLWA 33	37.224965	-95.031858
MLWA 35E	37.223696	-95.002268
MLWA 35W	37.225870	-95.013272
MLWA 36	37.244601	-95.037620
MLWA 38E	37.251762	-94.926703
MLWA 38W	37.248576	-94.940461
MLWA 39	37.252689	-94.984668
MLWA 40	37.264013	-94.976427
MLWA 41	37.261499	-94.958279
MLWA 42E	37.259492	-94.924293
MLWA 42W	37.257327	-94.936826
MLWA 44	37.267074	-94.934636
MLWA 45	37.283367	-94.912269
MLWA 4E	37.433128	-94.617333
MLWA 4W	37.438060	-94.630769
MLWA 5	37.411957	-94.768700
MLWA 6N	37.423991	-94.754964
MLWA 6S	37.415987	-94.758231
MLWA 7N	37.396332	-94.778641
MLWA 7S	37.388040	-94.783519
MLWA 8	37.389996	-94.772590
MLWA 9	37.287609	-94.772275
Monahan Outdoor Education Center	37.350972	-94.801386
Natural History Reserve	37.374343	-94.781406
Pittsburg Bike Park	37.428762	-94.693380
Pittsburg High School	37.409146	-94.670453
Pittsburg Industrial Park	37.433169	-94.683672
Pittsburg State University	37.391364	-94.697968
Stefanoni*	-	-
Wilderness Park	37.454764	-94.713891

Appendix II. Detections of nine anuran species heard calling from 65 sites. Detection at each site is indicated as the following: blank = not detected, 21 = only detected in 2021, 22 = only detected in 2022, and X = detected in 2021 and 2022.

Survey Point	American bullfrog	American toad	Blanchard's cricket frog	Boreal chorus frog	Cope's gray treefrog	Crawfish frog	Gray treefrog	Southern leopard frog	Spring peeper
Buche Wildlife Area	22	X	X	X	X	X	22	X	X
Ford N	X	22	X	X	X		X	X	
Ford S	X	X	X	X	X			X	
MLWA 10	X	X	X	21	X		21	X	X
MLWA 11	X	X	X	X	X			X	X
MLWA 12N	22	X	X	X	X			X	X
MLWA 12W	X	X	X	X	X		X	X	X
MLWA 13	X	X	X	X	X			X	X
MLWA 14	X	X	X	X	X	22	21	X	X
MLWA 16		X	X	X	X			X	X
MLWA 17S	X	X	X	X	X			X	
MLWA 17W	X	X	X	X	X			X	
MLWA 18E	X	X	X	X	X	22		X	
MLWA 18N	X	X	X	X	X			X	
MLWA 18S	X	X	X	X	X	X	21	X	
MLWA 19	X	X	X	X	X	22		X	
MLWA 1E	X	X	X	X	X			X	X
MLWA 1N	22	X	X	X	X	21	22	X	22
MLWA 1S		22	X	X	X		22	22	X
MLWA 21E	X	X	X	X	X			X	
MLWA 21S	X	X	X	X	X			X	
MLWA 21W	X	X	X	X	X		X	X	
MLWA 22E	X	X	X	X	X			X	
MLWA 22S	22	22	X	X	X		21	X	
MLWA 23	X	22	X	X	X			X	

MLWA 24E	X	X	X	X	X		22	X	
MLWA 24W	X	22	X	X	X			X	
MLWA 25	21	X	X	X	X			X	
MLWA 26	X	X	X	X	X			X	X
MLWA 27	X	X	X	X	X	X	22	X	
MLWA 28	X	X	X	X	X	X	X	X	
MLWA 29	X	22	X	X	X	X	22	X	
MLWA 3	X	X	X	X	X	21		X	X
MLWA 30	X	22	X	X	X		22	X	
MLWA 32	22	X	X	X	X			X	
MLWA 33	X	X	X	X	X		21	X	
MLWA 35E	X	X	X	X	X	X		X	
MLWA 35W	X	X	X	X	X	21	22	X	
MLWA 36	X	X	X	X	X			X	
MLWA 38E	X	X	X	X	X	X		X	
MLWA 38W	X	X	X	X	X	22		X	
MLWA 39	X	X	X	X	X		X	X	
MLWA 40	X	22	X	X	X	22	21	X	
MLWA 41	22		X	X	22			X	
MLWA 42E	X	X	X	X	X			X	
MLWA 42W	X	X	X	X	X			X	
MLWA 44	22	X	X	X	X	22		X	
MLWA 45	X	X	X	X	X			X	22
MLWA 4E	X	X	X	X	X			X	X
MLWA 4W	X	X	X	X	X	21		X	X
MLWA 5	X	X	X	X	X			X	X
MLWA 6N	X	X	X	X	X	X		X	X
MLWA 6S	X	X	X	X	X			X	X
MLWA 7N	X	X	X	X	X	21		X	X
MLWA 7S	X	X	X	X	X	21		X	22
MLWA 8	X	X	X	X	X	X		X	X
MLWA 9	22	X	X	X	X	21		X	X

Monahan Outdoor Education Center	22	22	X	X	X			X	X
Natural History Reserve	X	X	X	X	X	X		X	X
Pittsburg Bike Park		X	X	X	X		22	X	X
Pittsburg High School	X	X	X	X	X			X	X
Pittsburg Industrial Park		X	X	X	X	21	22	X	X
Pittsburg State University		22	X						
Stefanoni		X	X	X	X	21	22	X	X
Wilderness Park		X	X	X	X	21	22	X	X

Appendix III. Occupancy models estimating the probability that American bullfrogs (*Lithobates catesbeianus*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + obs) ψ(water + built)	8	0	0.92
p(day + obs) ψ(water + built + grass)	9	5.06	0.07
p(day + obs) ψ(built + watershed)	8	9.78	0.01
p(day + obs) ψ(water + built +watershed)	9	13.26	0
p(day + obs) ψ(built)	7	18.17	0
p(day + obs) ψ(built + wetland)	8	20.45	0
p(day + obs) ψ(built + forest)	8	20.45	0
p(day + obs) ψ(built + year)	8	21.62	0
p(day + obs) ψ(watershed)	7	23.60	0
p(day + obs) ψ(water + built + wetland)	9	29.29	0
p(day + obs) ψ(forest)	7	29.43	0
p(day + obs) ψ(crop)	7	31.80	0
p(day + obs) ψ(.)	6	32.35	0
p(day + obs) ψ(year)	7	32.73	0
p(day + obs) ψ(wetland)	7	34.48	0
p(day + obs) ψ(built + crop)	8	40.56	0
p(day + obs) ψ(grass)	7	47.88	0
p(day + obs) ψ(water)	7	47.89	0
p(day + obs) ψ(built + grass)	8	50.15	0
p(day + obs) ψ(water + built + year)	9	52.47	0
p(day + obs) ψ(water + built + crop)	9	52.47	0
p(day + obs) ψ(water + built + forest)	9	52.48	0
p(.) ψ(.)	2	322.31	0

Appendix IV. Occupancy models estimating the probability that crawfish frogs (*Lithobates areolatus*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + obs) ψ(crop + watershed + year)	9	0	0.65
p(day + obs) ψ(crop + watershed)	8	4.88	0.06
p(day + obs) ψ(crop + watershed + built)	9	5.74	0.04
p(day + obs) ψ(crop)	7	5.80	0.04
p(day + obs) ψ(crop + watershed + wetland)	9	5.86	0.03
p(day + obs) ψ(crop + watershed+ grass)	9	6.75	0.02
p(day + obs) ψ(crop + watershed + water)	9	6.81	0.02
p(day + obs) ψ(crop + watershed + forest)	9	7.18	0.02
p(day + obs) ψ(crop + wetland)	8	7.49	0.02
p(day + obs) ψ(watershed)	7	7.56	0.01
p(day + obs) ψ(crop + year)	8	7.59	0.01
p(day + obs) ψ(crop + grass)	8	7.74	0.01
p(day + obs) ψ(crop + built)	8	7.86	0.01
p(day + obs) ψ(water)	7	7.87	0.01
p(day + obs) ψ(crop + water)	8	7.98	0.01
p(day + obs) ψ(crop + forest)	8	8.01	0.01
p(day + obs) ψ(.)	6	9.23	0.01
p(day + obs) ψ(built)	7	10.07	0
p(day + obs) ψ(year)	7	10.43	0
p(day + obs) ψ(grass)	7	10.56	0
p(day + obs) ψ(forest)	7	10.85	0
p(day + obs) ψ(wetland)	7	11.02	0
p(.) ψ(.)	2	45.07	0

Appendix V. Occupancy models estimating the probability that gray treefrogs (*Hyla versicolor*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta \mathrm{AIC}_c$, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	$\Delta \mathrm{AIC}_c$	Model Weight
p(day + obs) ψ(grass)	7	0	0.3
p(day + obs) ψ(built)	7	0.42	0.24
p(day + obs) ψ(wetland)	7	1.65	0.13
p(day + obs) ψ(year)	7	2.62	0.08
p(day + obs) ψ(.)	6	2.76	0.08
p(day + obs) ψ(forest)	7	2.78	0.07
p(day + obs) ψ(water)	7	4.27	0.04
p(day + obs) ψ(crop)	7	4.61	0.03
p(day + obs) ψ(watershed)	7	4.73	0.03
p(.) ψ(.)	2	45.24	0

Appendix VI. Occupancy models estimating the probability that spring peepers (*Pseudacris crucifer*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + noise) ψ(watershed+ built + crop)	10	0	0.83
p(day + noise) ψ(watershed + built + water)	10	4.36	0.09
p(day + noise) ψ(watershed+ built + wetland)	10	7.23	0.02
p(day + noise) ψ(watershed + built)	9	7.81	0.02
p(day + noise) ψ(watershed+ built + forest)	10	8.53	0.01
p(day + noise) ψ(watershed + built + grass)	10	8.99	0.01
p(day + noise) ψ(watershed + built + year)	10	9.45	0.01
p(day + noise) ψ(watershed + forest)	9	10.12	0.01
p(day + noise) ψ(watershed + wetland)	9	12.82	0
p(day + noise) ψ(watershed + water)	9	13.48	0
p(day + noise) ψ(watershed + crop)	9	13.52	0
p(day + noise) ψ(watershed)	8	13.75	0
p(day + noise) ψ(watershed + grass)	9	13.93	0
p(day + noise) ψ(watershed + year)	9	15.39	0
p(day + noise) ψ(water)	8	76.26	0
p(day + noise) ψ(wetland)	8	89.20	0
p(day + noise) ψ(.)	7	97.95	0
p(day + noise) ψ(grass)	8	98.34	0
p(day + noise) ψ(built)	8	99.41	0
p(day + noise) ψ(year)	8	99.94	0
p(day + noise) ψ(crop)	8	100.22	0
p(day + noise) ψ(forest)	7	100.45	0
p(.) ψ(.)	2	210.19	0

Appendix VII. Site names, coordinates, and mining history for each survey site in southeast Kansas. Coordinates for sites on private property have been removed to protect the landowner's privacy.

Site	Latitude	Longitude	Mining History
Buche Wildlife Area†	37.31900	-94.68000	Non-mined
Buche Wildlife Area 2º	37.31967	-94.68082	Non-mined
Ford E*	-	-	Non-mined
Ford W*	-	-	Non-mined
MLWA 1	37.47519	-94.69988	Revegetated
MLWA 10	37.26732	-94.81289	Revegetated
MLWA 14	37.24484	-94.81422	Revegetated
MLWA 17	37.28233	-94.89190	Revegetated
MLWA 18	37.27416	-94.90721	Revegetated
MLWA 23 N	37.23625	-94.96997	Revegetated
MLWA 23 S	37.2305	-94.97710	Revegetated
MLWA 24	37.21294	-95.01171	Revegetated
MLWA 25	37.19983	-95.05648	Revegetated
MLWA 28	37.20794	-95.03116	Revegetated
MLWA 30	37.20951	-95.02092	Managed
MLWA 35	37.22534	-95.01129	Managed
MLWA 36	37.24368	-95.03973	Managed
MLWA 38	37.24885	-94.94020	Managed
MLWA 39	37.25316	-94.97762	Managed
MLWA 4 E	37.25583	-94.97166	Revegetated
MLWA 4 W	37.2681	-94.93485	Managed
MLWA 40	37.43318	-94.61997	Managed
MLWA 44	37.43891	-94.62923	Managed
MLWA 6 N	37.42294	-94.75732	Managed
MLWA 6 S	37.41605	-94.75536	Revegetated
MLWA 7	37.38795	-94.78133	Revegetated
Monahan Outdoor Education Center	37.34896	-94.80429	Managed
O'Malley Prairie	37.35270	-94.79471	Non-mined
Pittsburg High School	37.40999	-94.67033	Non-mined
Natural History Reserve	37.37444	-94.77864	Revegetated
Stefanoni*	-	-	Non-mined

† Site only surveyed in 2021
º Site only surveyed in 2022
* Site on private property

Appendix VIII. Amphibian species captured by dipnet and trapping at 31 sites from 2021 and 2022 in southeast Kansas. Captures are indicated as the following: blank = not captured, 21 = only captured in 2021, 22 = only captured in 2022, and X = captured in 2021 and 2022. Buche was only surveyed in 2021 and Buche 2 was only surveyed in 2022.

Common Name	**Scientific Name**	Buche2	Buche	Ford E	Ford W	HS	ML1	ML10	ML14	ML17	ML18	ML23 N
American bullfrog	*Lithobates catesbeianus*	22	21		x			x	x	x	22	22
American toad	*Anaxyrus americanus*	22		21	21				22			
Blanchard's cricket frog	*Acris blanchardi*	22	21	22	x	x	x	x	x	22	22	22
Boreal chorus frog	*Pseudacris maculata*			x	22		22					21
Crawfish frog	*Lithobates areolatus*	22		22	x							
Eastern newt	*Notophthalmus viridescens*											
Gray treefrog complex	*Hyla chrysoscelis/versicolor*			x	x		x		x			
Smallmouth salamander	*Ambystoma texanum*	22		21								
Southern leopard frog	*Lithobates sphenocephalus*	22		x	x		x	22	x		22	x
Spring peeper	*Pseudacris crucifer*	22					22		22			

Common Name	**Scientific Name**	ML23 S	ML24	ML25	ML28	ML30	ML35	ML36	ML38	ML39	ML4 E	ML4 W
American bullfrog	*Lithobates catesbeianus*	x		x		x		x	21	22	21	
American toad	*Anaxyrus americanus*							22				
Blanchard's cricket frog	*Acris blanchardi*	x	x	x	x	x		x	x		21	22
Boreal chorus frog	*Pseudacris maculata*	21	x			22		x				
Crawfish frog	*Lithobates areolatus*											
Eastern newt	*Notophthalmus viridescens*											
Gray treefrog complex	*Hyla chrysoscelis/versicolor*							x		22		
Smallmouth salamander	*Ambystoma texanum*							x				
Southern leopard frog	*Lithobates sphenocephalus*	x	22	x	x	x	22	x		x	x	22
Spring peeper	*Pseudacris crucifer*										22	

Common Name	**Scientific Name**	ML40	ML44	ML6 N	ML6 S	ML7	Monahan	O'Malley	Reserve	Stefanoni
American bullfrog	*Lithobates catesbeianus*	22			22	22		21		
American toad	*Anaxyrus americanus*							21		22
Blanchard's cricket frog	*Acris blanchardi*	x	x		x	x	22	22	22	x
Boreal chorus frog	*Pseudacris maculata*	21	21	22				x		x
Crawfish frog	*Lithobates areolatus*									
Eastern newt	*Notophthalmus viridescens*			x	x					
Gray treefrog complex	*Hyla chrysoscelis/versicolor*	x	22	22	21	x		22		x
Smallmouth salamander	*Ambystoma texanum*	x		x				21		x
Southern leopard frog	*Lithobates sphenocephalus*	x	x	x	x	x	x	x		x
Spring peeper	*Pseudacris crucifer*			22						22

Appendix IX. Occupancy models estimating the probability that American bullfrog (*Lithobates catesbeianus*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(type) ψ(.)	5	0	0.17
p(.) ψ(.)	3	0.58	0.13
p(type) ψ(pH)	6	1.09	0.10
p(type) ψ(hydro)	6	1.60	0.08
p(type) ψ(emveg)	6	1.64	0.08
p(type) ψ(area)	6	1.82	0.07
p(type) ψ(pH + hydro)	7	2.00	0.06
p(type) ψ(DO)	6	2.30	0.05
p(type) ψ(pH + emveg)	7	2.75	0.04
p(type) ψ(fish)	7	2.83	0.04
p(type) ψ(cond)	7	3.01	0.04
p(type) ψ(pH + area)	7	3.46	0.03
p(type) ψ(pH + fish)	8	3.83	0.03
p(type) ψ(pH +hydro + emveg)	8	4.26	0.02
p(type) ψ(pH +cond)	8	4.91	0.01
p(type) ψ(type)	7	5.02	0.01
p(type) ψ(year)	6	5.06	0.01
p(type) ψ(pH + hydro +fish)	9	5.25	0.01

Appendix X. Occupancy models estimating the probability that Blanchard's cricket frog (*Acris blanchardi*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(day) ψ(hydro)	5	0	0.36
p(day) ψ(hydro + area)	6	1.29	0.19
p(day) ψ(.)	4	2.45	0.11
p(day) ψ(hydro + site)	7	2.66	0.10
p(day) ψ(area)	5	2.84	0.09
p(day) ψ(hydro + fish)	7	3.48	0.06
p(day) ψ(hydro + cond)	7	4.90	0.03
p(day) ψ(type)	6	5.14	0.03
p(day) ψ(cond)	6	6.29	0.02
p(day) ψ(fish)	6	6.85	0.01
p(day) ψ(year)	5	7.16	0.01
p(day) ψ(emveg)	5	12.11	0
p(day) ψ(DO)	5	12.49	0
p(day) ψ(pH)	5	12.49	0
p(.) ψ(.)	3	21.61	0

Appendix XI. Occupancy models estimating the probability that boreal chorus frog (*Pseudacris maculata*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day) ψ(emveg)	4	0	0.46
p(day) ψ(emveg + DO)	5	1.40	0.23
p(day) ψ(emveg + fish)	6	3.29	0.09
p(day) ψ(DO)	4	3.61	0.08
p(day) ψ(emveg + type)	6	3.83	0.07
p(day) ψ(type)	5	5.47	0.03
p(day) ψ(fish)	5	7.13	0.01
p(day) ψ(.)	3	7.33	0.01
p(day) ψ(area)	4	8.01	0.01
p(day) ψ(hydro)	4	8.73	0.01
p(day) ψ(cond)	5	9.21	0
p(day) ψ(pH)	4	9.62	0
p(day) ψ(year)	4	10.70	0
p(.) ψ(.)	2	24.47	0

Appendix XII. Occupancy models estimating the probability that *Hyla* spp. (*Hyla chrysoscelis/versicolor*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(day) ψ(fish)	6	0	0.23
p(day) ψ(fish + cond)	8	1.02	0.14
p(day) ψ(fish + area)	7	1.69	0.10
p(day) ψ(fish + emveg)	7	1.95	0.09
p(day) ψ(fish + pH)	7	2.51	0.06
p(day) ψ(fish + hydro)	7	2.57	0.06
p(day) ψ(fish + cond + emveg)	9	2.93	0.05
p(day) ψ(fish + cond + hydro)	9	2.94	0.05
p(day) ψ(fish + cond + pH)	9	3.06	0.05
p(day) ψ(fish + type)	8	3.13	0.05
p(day) ψ(fish + area)	9	3.52	0.04
p(day) ψ(fish + cond + type)	10	4.14	0.03
p(day) ψ(.)	4	6.04	0.01
p(day) ψ(emveg)	5	6.42	0.01
p(day) ψ(pH)	5	6.93	0.01
p(day) ψ(area)	5	6.99	0.01
p(day) ψ(hydro)	5	7.54	0.01
p(day) ψ(DO)	5	8.13	0
p(day) ψ(cond)	6	8.24	0
p(day) ψ(year)	5	8.46	0
p(day) ψ(type)	6	8.54	0
p(.) ψ(.)	3	11.95	0

Appendix XIII. Occupancy models estimating the probability that southern leopard frog (*Lithobates sphenocephalus*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(type + day) ψ(hydro)	7	0	0.22
p(type + day) ψ(.)	6	0.85	0.14
p(type + day) ψ(fish)	8	2.14	0.08
p(type + day) ψ(hydro + area)	8	2.34	0.07
p(type + day) ψ(hydro + emveg)	8	2.61	0.06
p(type + day) ψ(hydro +DO)	8	2.65	0.06
p(type + day) ψ(hydro + pH)	8	2.67	0.06
p(type + day) ψ(emveg)	7	2.95	0.05
p(type + day) ψ(pH)	7	3.27	0.04
p(type + day) ψ(hydro + fish)	9	3.34	0.04
p(type + day) ψ(DO)	7	3.34	0.04
p(type + day) ψ(area)	7	3.40	0.04
p(type + day) ψ(year)	7	3.95	0.03
p(type + day) ψ(hydro + fish + DO)	10	5.49	0.01
p(type + day) ψ(hydro + fish + pH)	10	5.85	0.01
p(type + day) ψ(cond)	8	5.91	0.01
p(type + day) ψ(type)	8	6.01	0.01
p(type + day) ψ(hydro + fish + emveg)	10	6.20	0.01
p(type + day) ψ(hydro + fish + area)	10	6.22	0.01
p(.) ψ(.)	3	12.48	0

Appendix XIV. Candidate set of models of the effects of wetland characteristics on the amphibian species richness in wetlands across southeast Kansas during 2021 and 2022. Null models have also been included, along with each models' parameters (K), ΔAIC_c, and weights. See Table 2.1 for variable definitions.

Models	K	ΔAIC_c	Model Weight
fish +cond + emveg	7	0	0.41
fish + cond	6	0.69	0.29
fish + cond + do	7	0.88	0.26
fish + cond + type	8	5.14	0.03
fish + emveg	5	11.64	0
fish + type	6	12.48	0
fish	4	13.09	0
fish + do	5	14.32	0
cond	4	18.27	0
emveg	3	20.21	0
do	3	21.13	0
type	4	22.78	0
area	3	23.86	0
ph	3	25.14	0
null	2	26.85	0
year	3	27.88	0
hydro	3	28.89	0

Appendix XV. Candidate set of models of the effects of wetland characteristics on the Shannon diversity index of amphibian in wetlands across southeast Kansas during 2021 and 2022. Null models have also been included, along with each models' parameters (K), ΔAIC_c, and weights. See Table 2.1 for variable definitions.

Model	K	ΔAIC_c	Model Weight
cond + fish + hydro	7	0	0.75
cond + fish	6	3.51	0.13
cond + fish + area	7	5.67	0.04
cond	4	6.57	0.03
cond + hydro	5	7.31	0.02
cond + area	5	7.34	0.02
fish	4	11.79	0
area	3	12.61	0
hydro	3	12.87	0
null	2	13.23	0
year	3	13.49	0
ph	3	14.28	0
do	3	15.14	0
emveg	3	15.18	0
type	4	17.58	0